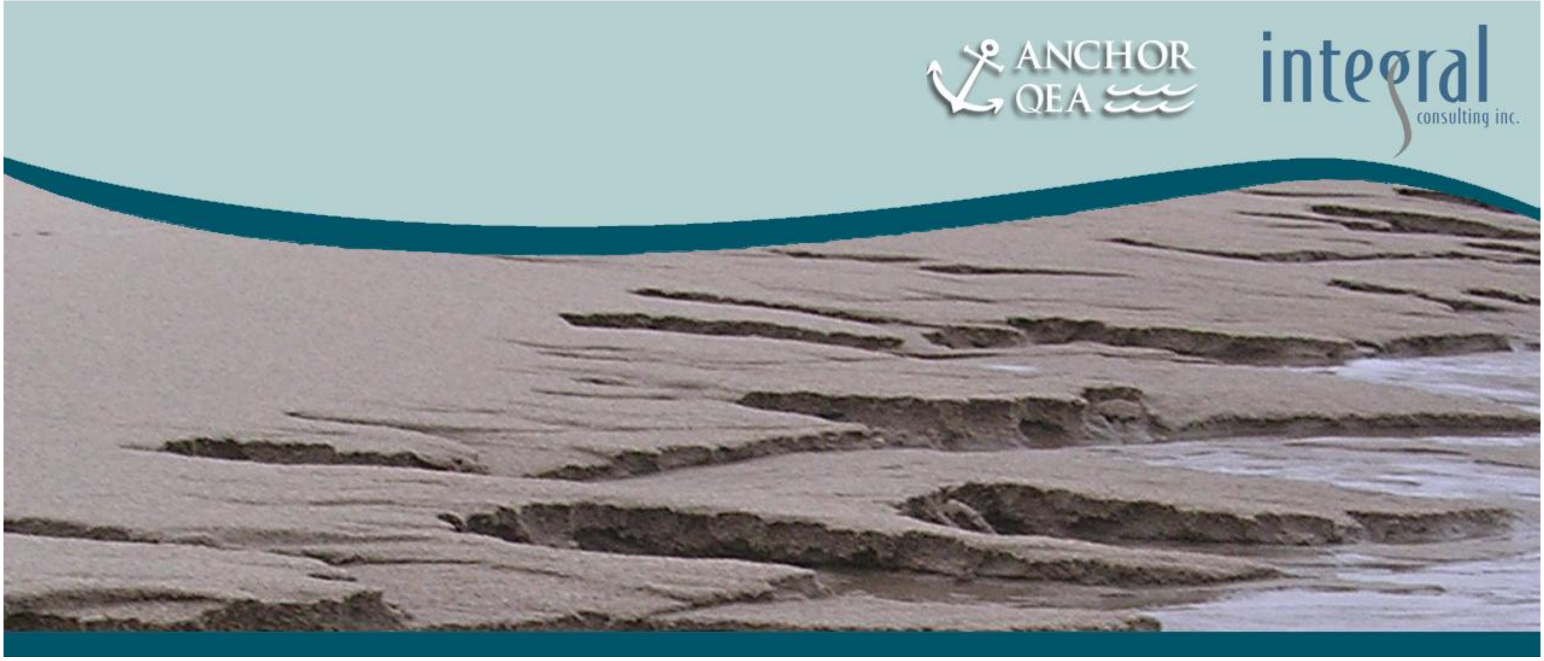

BASELINE HUMAN HEALTH RISK ASSESSMENT
SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

McGinnes Industrial Maintenance Corporation
International Paper Company
U.S. Environmental Protection Agency, Region 6

Prepared by

Integral Consulting Inc.
411 1st Avenue S, Suite 550
Seattle, Washington 98104

May 2013

BASELINE HUMAN HEALTH RISK ASSESSMENT

SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

McGinnes Industrial Maintenance Corporation

International Paper Company

U.S. Environmental Protection Agency, Region 6

Prepared by

Integral Consulting Inc.

411 First Avenue South, Suite 550

Seattle, Washington 98104

May 2013

TABLE OF CONTENTS

List of Tables

List of Figures

List of Appendices

LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
ABS_d	dermal absorption factor for soil/sediment
AhR	aryl hydrocarbon receptor
ATSDR	Agency for Toxic Substances and Disease Registry
BEHP	bis(2-ethylhexyl)phthalate
BHHRA	Baseline Human Health Risk Assessment
CSM	conceptual site model
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act of 1980
COI	chemical of interest
COPC	chemical of potential concern
$COPC_H$	chemical of potential concern for human health
CSF	cancer slope factor
CTE	central tendency exposure
CWA	Coastal Water Authority
DLC	dioxin-like compound
DMP	Data Management Plan
DQO	Data Quality Objective
EAM	Exposure Assessment Memorandum
EPC	exposure point concentration
FCA	fish collection area
HEAST	USEPA's Health Effects Assessment Summary Tables
HI	hazard index
HQ	hazard quotient
I-10	Interstate Highway 10
IPC	International Paper Company
IRIS	Integrated Risk Information System
JECFA	Joint Food and Agriculture Organization/World Health Organization Expert Committee on Food Additives
LADD	lifetime average daily dose

LOAEL	lowest-observed-adverse-effects level
MIMC	McGinnes Industrial Maintenance Corporation
NOAEL	no-observed-adverse-effects level
NTP	National Toxicology Program
OEHHA	Office of Environmental Health Hazard Assessment
PCB	polychlorinated biphenyl
PEF	particulate emission factor
POD	point of departure
PRA	probabilistic risk assessment
PSCR	Preliminary Site Characterization Report
RAGS	Risk Assessment Guidance for Superfund
RBA	relative bioavailability adjustment
REP	relative effect potencies
RfD	reference dose
RI	Remedial Investigation
RI/FS	Remedial Investigation and Feasibility Study
RME	reasonable maximum exposure
RsD	risk-specific dose
SAB	Science Advisory Board
SALG	Seafood and Aquatic Life Group
SAP	sampling and analysis plan
Site	San Jacinto River Waste Pits site in Harris County, Texas
SSL	soil screening level
TCDD	tetrachlorodibenzo-*p*-dioxin
TCRA	time-critical removal action
TDI	tolerable daily intake
TDSHS	Texas Department of State Health Services
TEF	toxicity equivalency factor
TEQ	toxicity equivalent
TEQ_{DF}	toxicity equivalent for dioxins and furans calculated using mammalian toxicity equivalency factors
TEQ_{P}	toxicity equivalent for polychlorinated biphenyls calculated using mammalian toxicity equivalency factors

TESM	Toxicological and Epidemiological Studies Memorandum
TMDL	total maximum daily load
UAO	Unilateral Administrative Order
UCL	upper confidence limit on the mean
USEPA	U.S. Environmental Protection Agency
WOE	weight-of-evidence

1 INTRODUCTION

This draft Baseline Human Health Risk Assessment (BHHRA) was prepared on behalf of International Paper Company (IPC) and McGinnes Industrial Maintenance Corporation (MIMC; collectively referred to as the Respondents) in fulfillment of the Unilateral Administrative Order (UAO), Docket No. 06-03-10, issued by the U.S. Environmental Protection Agency (USEPA) to IPC and MIMC on November 20, 2009 (USEPA 2009b), for the San Jacinto River Waste Pits site in Harris County, Texas (the Site).[1] The UAO directs the Respondents to perform a Remedial Investigation and Feasibility Study (RI/FS) which includes a BHHRA as part of the Remedial Investigation (RI). This document fulfills the UAO requirement for the BHHRA and also builds on the conceptual site models (CSMs) described in the Preliminary Site Characterization Report (PSCR) (Integral and Anchor QEA, 2012b) and the Exposure Assessment Memorandum (EAM) (Integral 2012a) for the area included within USEPA's Preliminary Site Perimeter[2]—the impoundments north of Interstate Highway 10 (I-10) and aquatic environment (Figure 1-1) and for the southern impoundment (Figure 1-2).

USEPA's Preliminary Site Perimeter (Figure 1-3), as presented in the UAO and discussed more fully in the RI Report and in Section 2.1 below, includes several impoundments used in the mid-1960s for the disposal of paper mill wastes and in-water and upland areas. The UAO made reference only to two impoundments located to the north of I-10. USEPA has subsequently required an investigation of an impoundment located on the peninsula to the south of I-10, citing historical documents that indicate possible waste disposal activities in that area.[3] In light of this, and in parallel with the organization of the RI Report, this BHHRA addresses these two impoundment areas separately, as the "northern impoundments" or "impoundments north of I-10" and as the "southern impoundment." Where appropriate, investigations and analyses that were performed separately in these two areas of study are differentiated in the text using references to the "area north of I-10" and

[1] References to "the Site" in this document are intended as reference to the formally designated SJRWP Superfund site and not to a geographical area.

[2] For the purposes of this document, the term "USEPA's Preliminary Site Perimeter" refers to the area shown within the "preliminary perimeter" in Appendix B of the UAO.

[3] The Respondents have submitted letters to USEPA dated July 20, 2011, setting out their respective positions with regard to the inclusion of the "southern impoundment" as a part of the RI/FS under the UAO.

the “area of investigation on the peninsula south of I-10”. The distinction between these areas primarily applies to information on hypothetical terrestrial exposure scenarios that involve possible human contact with upland soil. For organizational purposes, exposures and risks from contact with aquatic media (i.e., sediment and tissue) are presented together with the discussion of potential exposures and risks for the area north of I-10.

1.1 Purpose

USEPA guidance for conducting an RI/FS under the Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA) requires that remedies at contaminated sites be protective of human health and the environment (USEPA 1988). Baseline risk assessments evaluate the potential threats to human health and to the environment posed by sites in the absence of any remedial action. Specifically, a BHHRA is an analysis of the potential adverse health effects for individuals who may be exposed to or may be reasonably anticipated to be exposed in the future to hazardous substances released from a site in the absence of any actions to control or mitigate those releases. The results of the BHHRA are used to help determine whether remedial action is needed, and to provide the basis for the evaluation of the effectiveness of any subsequent remedial action. Specifically, results of the BHHRA provide a point of reference for evaluating risks under the no-action alternative and for quantifying risk reduction that can be achieved by each of the other remedial alternatives considered in the feasibility study. Risk models in the BHHRA are based on hypothetical exposure scenarios under baseline conditions, and are not intended to and cannot be utilized to determine whether any actual exposures are occurring or may have occurred. Because they are based on hypothetical exposure constructs, they also cannot be used to identify any actual adverse health effects from any exposures.

A description of baseline conditions and an overview of key aspects of the approach employed for this BHHRA are provided below. Each of these aspects is described in greater detail in subsequent sections of this BHHRA.

1.2 Baseline Conditions

For the area north of I-10 and aquatic environment, baseline specifically means environmental conditions that existed immediately prior to implementation of the time-critical removal action (TCRA). For the area of investigation on the peninsula south of I-10, baseline refers to the current condition. Baseline conditions are characterized for the BHHRA using the baseline dataset, as discussed further in Section 3.1. TCRA construction was completed in 2011 and involved installation of fencing and warning signs in addition to construction of an armored cap over the northern impoundments. The TCRA and the manner in which it changed potential human exposures are discussed further in Section 2.1. There is no basis for assuming that baseline represents conditions that existed at any time earlier than immediately prior to the TCRA, or that baseline conditions would have continued to exist had the TCRA not been implemented: data to describe conditions within USEPA's Preliminary Site Perimeter prior to initiation of the RI consisted of sediment and limited tissue data resulting from the TCEQ total maximum daily load (TMDL) monitoring program for dioxins and furans from 2000 to 2004, and the "intensive" sampling for dioxins and furans in sediments conducted in 2005 (University of Houston and Parsons 2006), as well as limited sampling reported in 1995 (ENSR and EHA 1995). These data sets are reviewed in the Sediment and Tissue SAPs (Integral and Anchor QEA 2010; Integral 2010b). Other than these data, pre-TCRA chemical and risk conditions within USEPA's Preliminary Site Perimeter are not described.

1.3 Overview of Approach

The approaches and methodologies presented in this BHHRA are consistent with USEPA guidance for conducting human health risk assessments and with data quality objectives (DQOs) and related statements and information presented by the sediment, tissue, and soil sampling and analysis plans (SAPs) that were submitted to and approved by USEPA (Integral and Anchor QEA 2010; Integral 2010a,b, 2011b,c), and the RI/FS Work Plan (Anchor QEA and Integral 2010). USEPA guidance that was considered for this BHHRA included, but was not limited to:

- Risk Assessment Guidance for Superfund (RAGS) Volume I Part A (USEPA 1989)
- RAGS Volume I Part B—Development of Risk-Based Preliminary Remediation Goals (USEPA 1991a)
- RAGS Volume I Part C—Risk Evaluation of Remedial Alternatives (USEPA 1991b)
- Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors (USEPA 1991c)

- Superfund's Standard Default Exposure Factors for the Central Tendency and Reasonable Maximum Exposure (USEPA 1993)
- Soil Screening Guidance: User's Guide (USEPA 1996)
- Exposure Factors Handbook (USEPA 2011a)[4]
- Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites (USEPA 2002c)
- RAGS Volume I Part E—Supplement Guidance for Dermal Risk Assessment (USEPA 2004)
- Texas Administrative Code sections containing exposure equations and parameters (TAC 350.74-75)

In line with the requirements in the UAO, an Exposure Assessment Memorandum (EAM) (Integral 2012a) and a Toxicological and Epidemiological Studies Memorandum (TESM) (Integral 2012b) were prepared and submitted to USEPA. These memoranda described the specific, hypothetical human use scenarios, exposure assumptions, and toxicological criteria to be used in this BHHRA. The final EAM and TESM are included as methodological appendices to this document (Appendix A and B, respectively).

Key aspects of the evaluation process for this BHHRA are summarized below, including identification of the chemicals of potential concern (COPCs), the hypothetical exposure scenarios evaluated, the types of potential health effects evaluated, the tiered approach used for selecting exposure scenarios for refined analyses, and the manner in which uncertainties in the risk assessment were addressed.

1.3.1 Chemicals of Potential Concern

Chemicals of potential concern for human health ($COPC_{HS}$) are selected in order to help focus a BHHRA on the chemicals that may drive human health risks.

[4] The RI/FS Work Plan (Anchor QEA and Integral 2010) prescribed the use of USEPA's 1997 *Exposure Factors Handbook* (USEPA 1997a) and USEPA's 2008 *Child Specific Exposure Factors Handbook* (2008). Since the publication of the RI/FS WP, EPA has updated its *Exposure Factors Handbook* (USEPA 2011a), which was used for the BHHRA.

The EAM and TESM presented COPC$_{HS}$ for the area north of I-10 and aquatic environment (Table 1-1). These COPC$_{HS}$ were identified according to steps described in the RI/FS Work Plan (Anchor QEA and Integral 2010) and the Sediment SAP (Integral and Anchor QEA 2010). Briefly, chemicals of interest (COIs) were identified as constituents that could have been associated with the paper mill waste deposited into the impoundments during the 1960s. COIs were further screened to identify COPC$_{HS}$. This screen considered comparisons with risk-based screening values, bioaccumulation potential, and whether or not the COI was detected in sediments from within the impoundment area. The selection of COPC$_{HS}$ for the area north of I-10 and aquatic environment is documented in Appendix C of the RI/FS Work Plan (Anchor QEA and Integral 2010) and in the COPC Technical Memorandum (Integral 2011a).

At the time the EAM and TESM were submitted, characterization of the soils in the area of investigation on the peninsula south of I-10 was ongoing; therefore, COPC$_{HS}$ for soils in this area were not presented in those documents. In May 2012, additional soil samples were collected from the area of investigation south of I-10 and analyzed for COIs. Data from the March 2011 Phase I soil sampling effort and the May 2012 Phase II investigation were screened to identify COPC$_{HS}$ for the area of investigation on the peninsula south of I-10 (Table 1-2). The methods and results of this screening are included as Appendices C and M to this document.

1.3.2 Human Exposure Scenarios Evaluated

The BHHRA characterizes the potential for adverse health effects to hypothetical receptors who may have used the Site under baseline conditions. As a result of TCRA implementation in 2011, the baseline condition no longer exists in the area north of I-10 and aquatic environment. For this area, the potential for adverse health effects to hypothetical receptors under the conditions following the TCRA (i.e., termed as the post-TCRA condition throughout this BHHRA) is also characterized.

As presented in the EAM, exposure media of concern for the area north of I-10 and aquatic environment are sediments and soils that hypothetical receptors may have contacted and fish and shellfish that may have been consumed. For the area north of I-10 and aquatic environment, potential health effects are quantified in this BHHRA using hypothetical recreational fisher, subsistence fisher, and recreational visitor scenarios. The risk evaluation was completed for a series of different hypothetical scenarios for each of these receptor groups. These scenarios assumed that an individual could have been exposed to different areas of the area north of I-10 and aquatic environment and/or could have ingested different types of tissue. Other hypothetical receptor groups who are assumed to have less contact with media in the area north of I-10 and aquatic environment than these receptors are qualitatively discussed within the context of these quantified results. For the area of investigation on the peninsula south of I-10, potential health effects were quantified for a hypothetical trespasser and worker.

1.3.3 Health Effects Evaluated

For this BHHRA, three categories of potential health effects were characterized. These were defined consistent with USEPA guidance as follows:

- Cancer risks—Defined as the incremental probability that an individual will develop cancer during his or her lifetime because of assumed exposure to a COPC at a site. The term "incremental" reflects the fact that the calculated risk associated with a site-related exposure is in addition to the background risk of cancer experienced by all individuals in the course of daily life. These risks were calculated for all potentially carcinogenic $COPC_{HS}$ that are assumed to have a linear dose response and no threshold dose.
- Noncancer hazards—The potential for noncancer health effects to occur was evaluated by comparing the estimated average daily intake of a chemical over the duration of assumed exposure to a toxicity criterion derived for a similar exposure period to calculate a hazard quotient (HQ) for each exposure route and $COPC_H$. HQs for multiple exposure routes evaluated for a single receptor group were summed to derive a $COPC_H$-specific hazard index (HI) for the receptor. HIs for compounds that cause toxicity at the same health endpoint were summed, resulting in a total HI for that receptor group. Unlike estimated cancer risks, the total HI is not a measure of probability, but instead is a measure of the likelihood and degree to which an adverse health effect might occur within the population evaluated (USEPA 1989).

- Dioxin cancer hazard—For some carcinogens a threshold (minimum) dose must be reached before a carcinogenic effect can occur. For these carcinogens, the potential for cancer to occur as a result of the assumed exposure is estimated using a hazard metric like that described for noncancer hazards above. The cancer hazard metric is used to evaluate dioxins and furans in this BHHRA. The use of this metric was established in the TESM, and is further discussed in Sections 5 and 6 below.

The manner in which each of these health effect metrics was interpreted is discussed in Section 5.

1.3.4 Tiered Approach for Risk Characterization

In this BHHRA, a tiered approach was applied for the risk characterization. A diagram outlining the approach used is provided as Figure 1-4. The three health effect categories described above were first evaluated for each potential receptor group and scenario via a deterministic evaluation. When the deterministic evaluation indicated that one or more of the following threshold criteria were met, additional evaluations to further characterize and refine the potential risks and/or hazards were considered for that scenario:

(1) The cumulative estimated exposure from all pathways resulted in an incremental cancer risk greater than one in 10,000 ($>1\times10^{-4}$)
(2) The cumulative estimated exposure from all pathways resulted in a total endpoint-specific noncancer HI>1
(3) The cumulative estimated exposure from all pathways resulted in a dioxin cancer HI>1.

For each scenario meeting one or more of these criteria, the refined analyses consists of three additional evaluations. These include 1) an analysis and comparison of background risks and/or hazards with the estimated deterministic risks and/or hazards for the area of study (i.e., either the area north of I-10 and aquatic environment or the area of investigation on the peninsula south of I-10), 2) an evaluation of post-TCRA risks and/or hazards, and 3) a probabilistic analysis of potential risks and/or hazards. Post-TCRA risks were only evaluated for scenarios and receptors considered by this BHHRA for the area north of I-10 and aquatic environment.

For the background evaluation, background risks and/or hazards for potential exposure routes included in the given scenario were calculated and compared to the deterministic risks and/or hazards for media being evaluated. This analysis allows for an evaluation of additional, incremental risk.

Risks and/or hazards for these potential exposure routes were also calculated for the post-TCRA condition. Post-TCRA risks and/or hazards were only calculated for dioxins and furans.[5] As outlined in the EAM for the Site, and described further in Section 2.1, the TCRA included capping that provided a barrier to direct contact with sediments in the northern impoundments and fencing that limited access to certain areas within USEPA's Preliminary Site Perimeter, including the capped area and surroundings (Figure 1-5). This comparison of potential baseline and post-TCRA risks and/or hazards allows the risk reduction achieved by the TCRA to be quantified.

In addition to the background and post-TCRA comparisons, any scenario that resulted in deterministic risk estimates that exceed one or more of the risk threshold criteria described above was evaluated using a probabilistic risk assessment (PRA). As is more fully discussed in Section 5.2.3.3, PRA uses probability distributions to characterize variability or uncertainty in exposure and risk estimates (USEPA 2001), and ultimately offers more detailed insight into both the magnitude and the probability of any potential exposure and risk. The PRA was performed for those $COPC_{HS}$ that contribute ≥5 percent of the overall risk and/or hazard in the selected scenarios, under the rationale that $COPC_{HS}$ that contributed ≥5 percent to the pathway-specific hazard/risk associated with a specific medium are considered potential risk drivers. The term "risk driver," which is repeated throughout this BHHRA, refers to these specific chemicals. Potential risks associated with the area under study and background risks and/or hazards were evaluated as part of the PRA.

[5] As is further described in Section 5.2.3.2, data for all $COPC_{HS}$ in all media of interest for post-TCRA conditions are not available and therefore, dioxins and furans were used to provide a relative measure of hazard and/or risk. Dioxins and furans have been established as an indicator chemical for the RI. Use of an appropriately chosen indicator chemical focuses the remedial strategy and is consistent with USEPA (1988) guidance for conducting an RI/FS under CERCLA.

1.3.5 Characterization of Uncertainty

There is uncertainty in the results of any risk assessment. USEPA (1989) guidance states the importance of presenting and discussing the uncertainties in the risk assessment in order to place the risk estimates in proper perspective. For this BHHRA, the sources of uncertainty and their overall impact on the risk results are discussed, with a focus on those uncertainties that impact the overall results to the greatest degree. Both quantitative and qualitative evaluations of uncertainty were completed, depending on the amount and type of information available.

1.4 Document Organization

This document is organized as follows:

- Section 2. Background
- Section 3. Hazard identification
- Section 4. Toxicity assessment
- Section 5. Exposure and risk characterization for the area north of I-10 and aquatic environment
- Section 6. Exposure and risk characterization for the area of investigation on the peninsula south of I-10
- Section 7. References.

It also includes the following appendices:

- Appendix A Exposure Assessment Memorandum
- Appendix B Toxicological and Epidemiological Studies Memorandum
- Appendix C Screening Analysis for Surface and Shallow Subsurface Soil in the Area of Investigation on the Peninsula South of I-10
- Appendix D Supplemental Toxicological and Chemical-Specific Parameters
- Appendix E Exposure Point Concentrations for Baseline and Background Exposure Estimates
- Appendix F Post-TCRA Exposures and Risks for the Area North of I-10 and Aquatic Environment: Methods and Results
- Appendix G Exposure Assumptions for Probabilistic Assessment

- Appendix H Human Health Exposure and Risk Estimates for the Area North of I-10 and Aquatic Environment
- Appendix I Human Health Exposure and Risk Estimates for Background
- Appendix J Human Health Exposure and Risk Estimates for the Area of Investigation on the Peninsula South of I-10
- Appendix K Human Health Risk Assessment for the Area North of I-10 and Aquatic Environment, Central Tendency Exposure Child Uncertainty Evaluation
- Appendix L Income Level and Ethnic/Cultural Background as Predictors of Fish Consumption Rates
- Appendix M Screening Analysis and Exposure Unit Identification for Evaluation of Soils 0 to 10 feet Deep, Area of Investigation on the Peninsula South of I-10
- Appendix N Response to USEPA Comments

2 BACKGROUND

This BHHRA draws on the findings of a number of studies and documents that have been submitted to and approved by USEPA (Integral 2012a,b; Anchor QEA and Integral 2010; Integral and Anchor QEA 2010; Integral 2010b, 2011a,b,c) and it provides a key component of the analyses required for the RI Report. This section briefly presents background information on the Site setting, population demographics, and receptor groups evaluated in this BHHRA.

2.1 Site Setting

USEPA's Preliminary Site Perimeter includes several impoundments that were used in the mid-1960s for the disposal of paper mill wastes, and in-water and upland areas as depicted in Figure 1-3.

The northern impoundments consist of two impoundments, together occupying approximately 14 acres, and are located on a 20-acre parcel north of the I-10 Bridge on the western bank of the San Jacinto River. Historical documents and aerial photographs suggest that in the mid-1960s an additional impoundment (i.e., the southern impoundment) was constructed on a peninsula of land south of I-10 and may have been used for the disposal of paper mill waste. At various times, the southern impoundment area and other portions of the area south of I-10 may have been used for the disposal of other waste material. Figure 1-3 shows the area within USEPA's Preliminary Site Perimeter, as presented in the UAO, and notes the specific area for the soil investigation south of I-10.

Implementation of a TCRA to address soils and sediments associated with the impoundments north of I-10 was completed in 2011. Through the installation of geotextile and geomembrane underlayments and a granular cover, the TCRA stabilized the entire area within the 1966 perimeter of the impoundments north of I-10 (Figure 2-1). Fencing installed as part of the TCRA implementation limited access to the impoundments north of I-10, areas to the immediate west of these impoundments, and the eastern shore of the San Jacinto River immediately adjacent to I-10. The Coastal Water Authority (CWA) also installed fencing on the east side of the San Jacinto River channel, along the western side of a road that passes under the I-10 Bridge, limiting access to the shoreline in this area. The placement of fences is shown in Figure 1-5. The condition that resulted from the TCRA and the additional fencing installed by the CWA collectively are described in this document as the "post-TCRA" condition.

2.2 Demographics

The area within USEPA's Preliminary Site Perimeter is located in Channelview, a suburb of Houston in Harris County, Texas. At the time of the 2010 census, the population of Harris County was 4,092,459, with 8.2 percent of the population under 5 years of age and 30.8 percent under the age of 20 years. Fifty-seven percent of the population was Caucasian, 19 percent African American, 6 percent Asian, with the remainder made up of individuals of another race or mixed race. Approximately 40 percent of individuals were Hispanic.[6] The median household income was $51,000. Approximately 17 percent of individuals and 14 percent of families had incomes below the national poverty level for one year or longer during the period from 2005 to 2010 (USCB 2012).

There are a number of surrounding communities from which individuals might come to visit the Site. The closest surrounding communities are Highlands and Baytown. While McNair is also close to USEPA's Preliminary Site Perimeter, this is an unincorporated area partly included in the Baytown census tract. Barrett and Crosby are located further upstream of USEPA's Preliminary Site Perimeter, on the eastern side of the San Jacinto River. Table 2-1 provides a comparison of the demographic characteristics of these communities.

There are some notable differences among these communities when compared with each other, and when compared with the demographic characteristics of Harris County and the State of Texas. While the median value of owner occupied housing units is higher in Harris County than in any of the identified communities or Texas statewide, the median household income for all towns except Barrett is similar to Harris County and higher than the median household income reported for the state. All of the identified communities have a lower per capita income than the county as a whole, with the largest difference for Barrett. The other notable difference is that the racial/ethnic demographics in Barrett differ from the

[6] Hispanic origin can be viewed as the heritage, nationality group, lineage, or country of birth of the person or the person's parents or ancestors before their arrival in the United States. People who identify their origin as Hispanic may be any race.

racial/ethnic demographics for the other identified communities, the county, and the state. This community has a substantially higher African American population and lower Caucasian and Hispanic populations than the other communities, the county or the state.

2.3 Conceptual Site Models

USEPA defines a CSM for site investigation as a written description and a visual representation of the predicted relationship between a stressor and a potential receptor (USEPA 1998) and it describes the potential sources, release mechanisms, transport pathways, and environmental exposure media of chemicals to receptors. The CSM provides a framework that facilitates application of the risk assessment process to the conditions and use of a site.

An exposure pathway links sources of COPCs to potential receptors and defines those links in terms of specific exposure routes. An exposure route is the physical way in which human receptors may come into contact with COPCs present in exposure media (i.e., ingestion, dermal absorption, inhalation). Under USEPA guidance, exposure pathways are considered potentially complete and significant if the potential exposure occurs frequently over an extended duration and/or the exposure medium represents a "significant" potential source of site-related COPCs to the receptor. Exposure pathways are considered potentially complete but "minor" if the exposure medium represents a relatively minor potential source of site-related exposure to a chemical, and/or potential for contact to the medium is limited. The relative importance of each pathway and route is relevant because pathways that are considered potentially complete and significant are those that provide the greatest risk reduction when addressed by remedial action.

Existing CSMs, developed in the RI/FS Work Plan (Anchor QEA and Integral 2010), refined in the PSCR (Integral and Anchor QEA 2012b), and following the draft of this BHHRA, describe the environment of the northern impoundments and aquatic environment and the area of investigation south of I-10 and the manner in which humans may have been exposed to impacted media in those areas under baseline conditions. It is important to understand these hypothetical receptors as models for the purposes of risk assessment and not as representing actual people. That is, the receptors indicated in the CSMs are constructs that were selected to represent the spectrum of potential hypothetical exposure intensities that could occur, for the purposes of full characterization of the range of possible exposure and risks. These CSMs are described below, with emphasis on the potentially complete and significant pathways and exposure routes.

2.3.1 Area North of I-10 and Aquatic Environment

The CSM for the area north of I-10 and aquatic environment is shown in Figure 1-1. Figure 2-2 identifies the potential routes of human exposure in detail and indicates whether they are considered significant or minor. For this area, hypothetical recreational and subsistence fishers, recreational visitors, and trespassers were identified as groups that may have contact with impacted media under baseline conditions. These hypothetical receptors represent a range of exposure types and intensities that could occur in the area north of I-10 and aquatic environment. For instance, the hypothetical subsistence fisher and hypothetical recreational fisher are assumed to be exposed to $COPC_{HS}$ via similar pathways. However, the hypothetical subsistence fisher is assumed to frequent the area more often, and consume a larger number of fish and shellfish from the area under evaluation compared to the hypothetical recreational fisher. The hypothetical recreational visitor and hypothetical trespasser are also assumed to be exposed to $COPC_{HS}$ via the same exposure pathways as one another; however, the hypothetical recreational visitor is assumed to be exposed at a greater frequency and duration relative to the trespasser. These receptor groups are discussed below following a general discussion of the minor pathways.

Consistent with the Public Health Assessment for the Site (TDSHS 2012), potential inhalation of $COPC_{HS}$ in air and exposure via direct contact with surface water were defined as minor pathways for this risk assessment. Inhalation exposure via vapor inhalation is considered minor because none of the $COPC_{HS}$ identified are volatile compounds and, therefore, would not tend to volatilize into ambient air. While inhalation of particulates derived from the resuspension of surface soil may occur, this pathway generally contributes less than one percent of total estimated exposure when direct soil contact pathways (ingestion and dermal contact) are considered. This is demonstrated with standard exposure assumptions used for determining residential and industrial soil screening levels (USEPA 2012b). Exposure to $COPC_{HS}$ in surface water is also considered to be a minor pathway for this site. This is because the primary COPCHS, dioxins and furans, are hydrophobic, are not soluble in water, and tend to be tightly bound to the organic carbon fraction of sediments.[7] It is possible that individuals could be exposed to COPCHS that adsorb to suspended sediment particles in the water column, but those exposures would be brief and minimal because the movement of the surface water will continually wash away the majority of the sediment particles that contact the skin, leaving little opportunity for absorption.

As described in the EAM (Appendix A) and the RI/FS Work Plan (Anchor QEA and Integral 2010), minor pathways were not evaluated quantitatively, but rather were addressed qualitatively. Specifically, information about the physical-chemical properties of the COPCHS defined as risk drivers were used to describe the likely extent of their presence in media for which exposures are considered minor. Evaluation of minor pathways also included a description of the likelihood, frequency, and intensity with which exposures via minor pathways and routes are anticipated to occur for each potential receptor.

2.3.1.1 Fishers

Fishing activity within the waters surrounding USEPA's Preliminary Site Perimeter has been observed and fishers in this area have been reported to collect whatever they catch (Beauchamp 2010, Pers. Comm.). However, little information is available about the type and amount of fishing that occurs. The limited information that is available is based on observations of the area within USEPA's Preliminary Site Perimeter. Specifically, fishing is reported to have been popular at the northern tip and along the northeast side of the area of the northern impoundments prior to implementation of the TCRA. People were observed to wade out in the water on the east side and fish and use crab cages in this area. Prior to implementation of the TCRA, fishing was reportedly also observed to the south of the northern impoundments area and under the I-10 Bridge, on both sides of the channel. Other points of fishing access within USEPA's Preliminary Site Perimeter include RV trailer parks on the east side of the river north of I-10 that provide access to the river, and a public access area at Meadowbrook Park to the west (Beauchamp 2010, Pers. Comm.).

[7] Available at http://www.epa.gov/ogwdw/pdfs/factsheets/soc/tech/dioxin.pdf

Fishers may potentially be exposed to $COPC_{HS}$ via direct contact with sediments and soils, and by ingesting fish or shellfish that have been exposed to impacted media. They may also potentially be exposed to $COPC_{HS}$ through direct contact with surface water (ingestion and dermal contact) and through inhalation of $COPC_{HS}$ as particulates or vapors in air; however, exposures via these media and routes are considered to be minor (Figure 2-2).

2.3.1.2 Recreational Visitors

Although the lands within USEPA's Preliminary Site Perimeter are largely privately owned, points of access were available to the public along and within this area under baseline conditions (i.e., immediately prior to the TCRA). Such access allowed for a variety of recreational activities other than fishing, including picnicking, walking, bird watching, wading, and boating. Shoreline use and wading within USEPA's Preliminary Site Perimeter were reportedly observed under baseline conditions (Beauchamp 2010, Pers. Comm.).

Recreational visitors could potentially be exposed via the same direct contact exposure routes as fishers (i.e., incidental ingestion of and dermal contact with soils and sediments). However, these individuals are not exposed via ingestion of fish or shellfish.

2.3.1.3 Trespasser

Signs of trespassing have been reported in some areas within USEPA's Preliminary Site Perimeter, particularly under the I-10 Bridge. Consistent with the hypothetical receptors addressed by the HHRA representing a spectrum of potential assumed exposures, the hypothetical trespasser is the receptor used to represent a very low level of possible exposure. Therefore, although a hypothetical trespasser could be exposed via the same pathways as the recreational visitor (i.e., direct contact pathways) and recreational fisher (i.e., ingestion of fish and shellfish), the concept of the trespasser is that of a person whose exposure would likely be intermittent and of a shorter term than the exposures being evaluated for either of those scenarios. Thus, for the area north of I-10, the estimated risks and hazards presented for the hypothetical fishers and hypothetical recreational visitors are higher than and would overstate potential risks for hypothetical trespassers. For this reason, and as discussed in the EAM, the hypothetical trespasser scenario was not evaluated quantitatively for the area north of I-10 and aquatic environment. A discussion of the exposure that would be anticipated for a hypothetical trespasser relative to exposures calculated for the recreational visitor and recreational fisher is, however, included as part of the risk characterization for the area north of I-10 and aquatic environment.

2.3.2 Area of Investigation South of I-10

The CSM for the area of investigation on the peninsula south of I-10 is shown in Figure 1-2. Figure 2-3 describes the specific routes of potential exposure in detail. For this area, trespassers and commercial workers were identified as groups that may potentially come into contact with impacted media. In comment number 7 on the draft of this BHHRA (Appendix N), USEPA requested that soils greater than 2 feet deep additionally be evaluated in the BHHRA. In response to this comment future construction workers were additionally evaluated in the BHHRA. These receptor groups are discussed below.

2.3.2.1 Trespasser

With signs of trespassing in areas along the western bank of the River within USEPA's Preliminary Site Perimeter, it is possible that trespassers might walk around or spend time in the area of investigation on the peninsula south of I-10. Because such activities might result in direct contact with surface soil, potentially complete exposure pathways for the trespasser are incidental ingestion and dermal contact with soil. Because fencing and active management and use of industrial properties south of I-10 make this area largely inaccessible, it is anticipated that the trespasser's exposure would be infrequent (i.e., an average of 24 times throughout the year). Also it is likely that trespassing activities by any given individual would be limited to a relatively short time frame (i.e., no more than a few years).[8]

2.3.2.2 Commercial Worker

Land use on the peninsula south of I-10 is commercial/industrial. Commercial workers, who perform maintenance or other work-related outdoor activities, might have potential direct contact with surface and shallow subsurface soil. Potentially complete exposure pathways

[8] As described in Section 2.3.1.3 for the hypothetical receptors for the area north of I-10 and aquatic environment, the trespasser is anticipated to visit the area with less frequency, and for a shorter duration, than a recreational visitor.

for the commercial worker are incidental ingestion and dermal contact with surface and shallow subsurface soil.

2.3.2.3 Construction Worker

In the future, construction work could occur in the area of investigation on the peninsula south of I-10. Under this future scenario, hypothetical construction workers might have direct contact with surface and subsurface soil. Potentially complete exposure pathways for the construction worker are incidental ingestion and dermal contact with surface and subsurface soils.

3 HAZARD IDENTIFICATION

Hazard identification consists of a data evaluation step to define appropriate environmental data relevant to potential human exposures. This section presents an overview of the data that were used to evaluate potential risks to under the scenarios evaluated and the data treatment rules that were applied.

3.1 Baseline Data

Available data used in this BHHRA to evaluate potential exposures are summarized in Table 3-1 and discussed below. This section describes the datasets used to assess potential exposures for the area north of I-10 and aquatic environment and the area of investigation on the peninsula south of I-10 and background exposures, and is followed by a description of the data types that were used. The specific data that were used to evaluate each potential exposure pathway under each exposure scenario are described in the EAM (Appendix A) and Section 5 of this BHHRA in the context of the individual potential receptor groups evaluated.

3.1.1 Datasets

The RI/FS Work Plan (Anchor QEA and Integral 2010) described the rationale for selection of data to be used in the baseline risk assessments. Data to be used in baseline risk assessments should be of known quality, which includes only Category 1 data (as described in Section 3 of the RI/FS Work Plan), and should reflect recent but pre-remediation (baseline) conditions. Based on a temporal analysis of surface sediment data in the area around the northern impoundments (Integral 2011a) and as established in the PSCR (Integral and Anchor QEA 2012b), data collected in 2005 or earlier are not considered reflective of recent conditions and were not considered representative of baseline conditions for purposes of this BHHRA.

Data from within USEPA's Preliminary Site Perimeter and background data were used in the risk assessment. Analysis of background information allows for consideration of other potential sources of COPCHS, and is relevant for the evaluation of remedial alternatives and for risk management decisions at the Site.

The baseline dataset for the BHHRA consists of:

- Sediment, tissue, and soil data collected for the RI/FS.
- Sediment and surface water data collected by URS (2010) for TCEQ in 2009.
- Polychlorinated biphenyl (PCB) congener data for fish tissue and sediments collected by TCEQ in 2008 and 2009 as part of the total maximum daily load (TMDL) program (University of Houston and Parsons 2009; Koenig 2010, Pers. Comm.)[9]

The background dataset consists of:

- Sediment, tissue, and soil data collected for the RI/FS in background areas.
  - Sediment—Sediment from 10 intertidal locations upstream from the upper boundary of USEPA's Preliminary Site Perimeter. Subtidal sediment samples from upstream were not used in this BHHRA.
  - Tissue—Edible crab and catfish tissue from Cedar Bayou and from fish collection area (FCA) 5 in the San Jacinto River estuary south of the Fred Hartman Bridge. Clams were collected along two sections of shoreline upstream of the upper boundary of USEPA's Preliminary Site Perimeter, downstream of the mouth of the San Jacinto River.
  - Soil—Soil from locations in two general areas; Burnet Park and the I-10 Beltway 8 Green Space.
- PCB congener data collected by TCEQ in 2008 and 2009 as part of the TMDL program from stations downstream of USEPA's Preliminary Site Perimeter and in proximity to the Fred Hartman Bridge (University of Houston and Parsons 2009; Koenig 2010, Pers. Comm.).

A comprehensive discussion of background data is included in the RI Report (Integral and Anchor QEA 2012a).

[9] Appendix A to the EAM (Integral 2012a) documents Integral's independent validation of TCEQ's PCB congener data according to procedures applicable to the RI/FS. This validation effort resulted in a change to the classification of these PCB data from Category 2 to Category 1.

3.1.2 Data Types

Data used in a BHHRA should represent conditions in environmental media that human receptors could potentially contact. The data types used to characterize each medium of interest are briefly discussed below. This information was presented in the EAM (Appendix A), and is summarized here for completeness.

3.1.2.1 Sediment

Fishers and recreational visitors may have the potential to be exposed to surface sediment in accessible shoreline areas within USEPA's Preliminary Site Perimeter. There is a limit to the water depth into which these individuals would wade during these activities. To determine the boundary of the sediment that might result in direct contact exposures, bathymetry contours were mapped. The 2-foot depth contour (i.e., sediment covered by 2 feet or less of water) was considered the outer boundary of sediments that people would contact directly.[10] All shoreline and nearshore sediment data covered by 2 feet or less of water were used to evaluate exposure to sediment for the fishing and recreational scenarios. As outlined in the Sediment SAP (Integral and Anchor QEA 2010) and EAM (Integral 2012a), sediment samples collected from the 0- to 6-inch depth increment were used to evaluate exposure to humans.

3.1.2.2 Tissue

The tissues collected under baseline conditions to evaluate potential human exposures (Integral 2010b) included hardhead catfish fillet (skin removed), edible crab tissue, and edible clam tissue. Hardhead catfish fillet data were used to estimate exposures resulting from the ingestion of finfish. Edible crab and clam tissues were used to estimate exposures via shellfish ingestion.

3.1.2.2.1 Fish Tissue Representativeness

There is uncertainty regarding the representativeness of available fish tissue data for characterizing potential exposures via ingestion that could have occurred under baseline conditions. There is no information regarding the extent to which various fish and shellfish

[10] The tidal condition at which the 0 foot contour was established is not known. This results in some uncertainty in the determination of sediment locations that are representative of human exposure.

types are collected from within USEPA's Preliminary Site Perimeter and consumed. In comment number 6 on the draft of this BHHRA (Appendix N), USEPA requested that any detail available on the types and sizes of fish that may have been captured and could be eaten by anglers within USEPA's Preliminary Site Perimeter be provided. However, there are no such data to describe the species preferences of anglers who use the area within USEPA's Preliminary Site Perimeter.

In addition, USEPA comment number 5 (Appendix N) indicates that the uncertainty arising from the absence of information on angler preferences is to be addressed using conservative assumptions. On the basis of information available at the time the Tissue SAP was prepared, hardhead catfish does provide a conservative representation of edible fish tissue for this risk evaluation. In preparation of the Tissue SAP, available tissue chemistry data for hardhead catfish, blue crab, and blue catfish collected from within and outside of USEPA's Preliminary Site Perimeter were evaluated. For the two catfish species, the mean, minimum, and maximum TEQ_{DF} concentrations were higher in hardhead catfish fillet from within USEPA's Preliminary Site Perimeter than in blue catfish fillet from the same area. The mean TEQ_{DF} concentration in hardhead catfish fillet was also higher within USEPA's Preliminary Site Perimeter than the mean of all hardhead catfish samples outside of it. Total PCB concentrations (as the sum of Aroclors) in fillet tissue of both catfish species were similar within USEPA's Preliminary Site Perimeter, and were similar to crab tissue, but were more variable (i.e., no pattern was evident) outside of USEPA's Preliminary Site Perimeter. In this context, hardhead catfish appeared to provide the most conservative estimate of TEQ_{DF} accumulation in edible fish tissue on the basis of the data available at the time.

This choice was further supported by a qualitative review of data for TEQ_{DF} in edible tissue of a broader range of fish species caught by TCEQ and TDSHS outside of USEPA's Preliminary Site Perimeter that were available at the time the Tissue SAP was prepared. A table of those data appears in Appendix B of the EAM (Integral 2012a), and includes TEQ_{DF} concentrations in edible tissues of blue catfish, blue crab, striped bass (*Morone saxatilis*), red drum (*Sciaenops ocellatus*), southern flounder (*Paralichthys lethostigma*), spotted seatrout (*Cynoscion nebulosus*), and hardhead catfish. The EAM is included as Appendix A of this document. Finally, USEPA (2009a) concluded that benthic fish species generally have higher tissue concentrations of dioxins and furans than predators in the same ecosystem, and hardhead catfish are benthic feeders (USFWS 1982; Yanez-Arancibia and Lara-Dominguez 1988; USFWS 1983). Therefore, on the basis of information on the fish collected from within USEPA's Preliminary Site Perimeter, collected within the Houston area, and collected in USEPA's national study of lake-dwelling fish (USEPA 2009a), use of hardhead catfish to represent all human exposure to finfish results in a conservative upper-end exposure for fishers consuming finfish.

3.1.2.2.2 Chemical Concentrations in Fish Tissue Relative to Fish Age

USEPA comment 5 on the draft of this BHHRA (Appendix N) also expresses concern that chemical concentrations may increase with fish age, and that the absence of information on ages of fish analyzed and age-preferences of anglers results in uncertainty regarding exposures to $COPC_{HS}$ via fish ingestion. In the case that documentation cannot be provided to support the assumption that the fish tissue analyzed is representative of the ages of the fish likely to be consumed, comment 5 requests "a credible projection of contaminants in mature catfish".

The Tissue SAP, which was reviewed and revised in response to USEPA comments and which was approved by USEPA, did not include a component to collect data on fish age. Because multiple fish were composited to form each fillet sample, each sample theoretically represents multiple fish ages. For these reasons, there is no way to estimate tissue concentrations on the basis of any age-concentration relationship that might be available. Moreover, although some research has shown that methylmercury can accumulate in fish tissue over time, resulting in a correlation between fish age and mercury concentrations in tissue (e.g., Lange et al. 1993; Grieb et al. 1990), demonstrations of such a relationship for PCBs are less common than for mercury, and such demonstrations for dioxins and furans were not found. One experimental study was found (Wang and Lee 2010) in which concentrations of PCBs and dioxins and furans in orange-spotted grouper (*Epinephelus coioides*) were monitored from hatch to 3 years of age.

Wang and Lee (2010) exposed orange-spotted grouper to dioxin and furan congeners and to dioxin-like PCB congeners in a controlled experiment, and monitored tissue concentrations from hatchlings for 36 months, resulting in congener concentration data for five separate ages. The wet-weight concentrations of each PCB congener, and both lipid-normalized and wet weight concentrations of the sum of PCB congeners and the TEQ_P, increase with fish age. This trend was not observed for dioxins and furans to the same extent as it was for PCBs. Wet-weight concentrations of TCDD and, to a lesser degree, TCDF increased in the orange-spotted grouper with age, but lipid-normalized concentrations of these congeners were unchanged with fish age. Wet-weight concentrations of all other dioxin and furan congeners, and the sum of dioxin and furan congeners, did not increase with age, and in some cases decreased with fish age. The lipid-normalized total PCDD and total PCDF concentrations both decreased with fish age. Other experimental studies addressing age-concentration relationships for dioxins and furans were not found.

Field studies are considered less reliable with respect to the question of age-concentration relationships because they use wild fish with uncertain and uncontrolled exposures. Although field studies have been published on this topic (e.g., Roots and Zitko 2006; Pandelova et al. 2008), their findings are equivocal with respect to the question of age-related increases in concentrations of dioxins and furans in fish tissue. Therefore, on the basis of Wang and Lee (2010), it appears that there may be some potential for PCB concentrations to increase with fish age, but concentrations of dioxins and furans are not expected to increase with fish age. This is consistent with the findings of the *Technical Memorandum on Bioaccumulation Modeling* (Integral 2010c), that dioxins and furans have limited potential for bioaccumulation and biomagnification in fish and benthic invertebrates because there are biological limits on uptake and because fish and invertebrates can metabolize and excrete dioxins and furans to an extent that varies for the different congeners.

Finally, the selection of hardhead catfish for the tissue study was consistent with protocols described by the TDSHS (2007) Quality Assurance Project Plan for their tissue monitoring program. That document indicates that hardhead catfish are a suitable estuarine fish species for tissue chemistry monitoring. The TDSHS methods do not specify that fish age data be collected and, in the case of hardhead catfish, do not indicate a fish length limit. The length limit given in the TDSHS methods for the other estuarine fish species considered suitable for monitoring is typically 12 inches (305 mm) or greater. Hardhead catfish collected during the 2010 tissue study for the RI ranged from 11.8 to 15.7 inches (300 to 400 mm). In their study of catfish, including the hardhead catfish, in the southern Gulf of Mexico, Yanez-Arancibia and Lara-Dominguez (1988) do not report capturing any hardhead catfish greater than 355 mm. Therefore, the fish tissue data used for the RI are consistent with fish tissue data used by TDSHS for monitoring chemical contamination of edible tissues, and size range targeted by the fish study encompassed the maximum reported by other scientists. Although fish size is not a direct measure of age, the tissue study design is expected to have resulted in capture of a random assortment of catfish in the largest size category, of which some are among the oldest in the population available to anglers.

In light of this information, there is no basis for concern that the tissue study could have resulted in a downward bias in the exposure assessment for dioxins and furans. In fact, the tissue study was more likely to have resulted in an upward bias in the human exposure assessment because hardhead catfish are a benthic fish (USEPA 2009a), and had been demonstrated to have higher TEQ_{DF} concentrations than other species captured within USEPA's Preliminary Site Perimeter.

Uncertainties associated with the representativeness of tissue data designated for this BHHRA and the likelihood of consumption of this species alone are explored in the uncertainty evaluation completed as part of the risk characterization (Section 5).

3.1.2.3 Soil

Fishers, recreational visitors, and trespassers have potential for exposure to $COPC_{HS}$ in soils in the impoundments north of I-10, while trespassers and workers may be exposed to $COPC_{HS}$ in soils in the area of investigation south of I-10. Fishers, recreational visitors, and trespassers are anticipated to participate in activities that would potentially bring them into contact only with surface soils. Workers, however, may have contact with a combination of surface and shallow subsurface soils during outdoor maintenance activities. Under the soil investigations completed for the RI, soil from a variety of depth increments was collected at various locations (Integral 2011b,c).

Soils representing the surface condition (i.e., those collected from surface increments of 0 to 6, 0 to 8, 0 to 12, and 0 to 24 inches) were used to evaluate potential exposure for fishers, recreational visitors, and trespassers. For commercial workers in the area of investigation on the peninsula south of I-10, data from these increments, as well as from the shallow subsurface increment of 6 to 12 inches, are used in the exposure evaluation.

Soils representing conditions in surface and deeper soils (i.e., those collected between surface and 10 feet) were used to evaluate potential exposure for future construction workers on the peninsula south of I-10. Soil samples from within the area of investigation south of I-10 were collected in the following increments: 0 to 6, 6 to 12, and 12 to 24 inches, and in 2-foot increments at depths greater than 2 feet. VOCs, SVOCs and PCBs were analyzed in every other increment deeper than 2 feet. All data for samples collected within the upper 10 feet of soils are used in the exposure evaluation.

3.2 Data Treatment

RI/FS data are managed according to the project Data Management Plan (DMP), which is Appendix A to the RI/FS Work Plan (Integral and Anchor QEA 2012b). For performance of various analyses in this BHHRA, general data treatment rules are as follows:

- 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) toxicity equivalent (TEQ) concentrations for dioxins and furans (i.e., as TEQ_{DF}) and PCBs (i.e., as TEQ_{P}) were calculated using the toxicity equivalency factors (TEFs) for mammals (Table 3-2) (Van den Berg et al. 2006; USEPA 2010e).
- TEQ concentrations in samples for which one or more dioxin-like congener was not detected were calculated in two ways. Under the first approach, censored data (i.e., nondetects) were assumed to be equal to one-half of the estimated detection limit for each congener. Under the second approach, nondetects were assigned a value of zero.
- Total PCBs in tissue were calculated as the sum of the 43 PCB congeners listed in Table 3-3. In cases in which additional PCB congeners co-eluted with the 43 specified congeners, these additional congeners were included in the summing to derive the total PCB concentration.
- In soil samples in which one or more Aroclor was detected, total PCBs were calculated as the sum of detected Aroclor concentrations only. When no Aroclors were detected, total PCBs for each sample was estimated at one-half the maximum

detection limit among all Aroclors in the sample.[11] This rule was not applied to the calculation of total PCBs in sediment because of elevated detection limits in these samples. The treatment of total PCBs in sediment is discussed further below.

- For TEQ and total PCB metrics, if the concentration of one or more individual constituent (i.e., congener or Aroclor) included in the summation was an estimated value, then the summed total was reported as estimated (J-qualified). If all constituents were not detected in a sample, then the summed concentration was reported as not detected (U–qualified). If one or more constituent was not detected, then the resulting total estimate was reported as estimated (J-qualified).
- One hundred percent of mercury detected in tissue was assumed to be methylmercury. For soil and sediment, it was assumed that 100 percent of mercury detected was an inorganic form.[12]
- Ten percent of arsenic detected in tissue was assumed to be inorganic arsenic. The remaining 90 percent was assumed to be in an organic form.[13] One hundred percent of the arsenic measured in soils and sediments was assumed to be inorganic arsenic.
- Any nondetects for a given analyte and medium that were higher than the maximum detected concentration for the same analyte and medium were considered "high-biasing non-detects," and were removed prior to use of the dataset in this BHHRA, as outlined in USEPA (1989) guidance.

The data treatment rule described above for calculation of total PCBs as Aroclors (i.e., calculated as the sum of detected Aroclors or as one-half of the highest detection limit among Aroclors when no Aroclors were detected) was not applied to estimate total PCBs for sediment because of analytical uncertainty for that dataset. Both Aroclors and dioxin-like PCB congeners were analyzed in sediment samples collected for the RI, consistent with the

[11] This approach is consistent with methods used in a recent BHHRA for the Lower Duwamish Waterway, in Seattle, Washington. PCBs are a COC for that Site. This BHHRA was approved by USEPA in 2007 (Windward 2007).

[12] These treatments are consistent with USEPA guidance (2010b) and the approaches taken by the Texas Department of State Health Services (TDSHS) Seafood and Aquatic Life Group (SALG) (TDSHS 2008).

[13] This treatment is consistent with the state of knowledge regarding the proportions of inorganic and organic arsenic in fish tissues (USEPA 2003b; ATSDR 2007) and approaches taken by TDSHS's SALG (TDSHS 2008).

Sediment SAP (Integral and Anchor QEA 2010).[14] In the analysis of some of the sediment samples collected for the RI from within the 1966 perimeter of the northern impoundments (including core samples), matrix interference resulted in elevated detection limits for Aroclors. Among all of the sediment samples in the 1966 perimeter, Aroclors were only detected in one sample, including those with matrix interferences. This single estimated (J-qualified) concentration of 1,400 µg/kg was for Aroclor 1254 in a subsurface (2-4 feet) sediment sample collected during the RI at station SJGB014. This estimated concentration was lower than the elevated detection limit for this Aroclor in two of the stations where matrix interferences occurred and detection limits were elevated, but much higher than nondetects in the same core with normal detection limits. Because this sample provided the only indication of Aroclors in sediments within USEPA's Preliminary Site Perimeter, and sediment EPCs for total PCBs were needed for the risk assessment, the sediment EPC for total PCBs was conservatively estimated as one-half the detection limit for Aroclor 1254 in each sample, with all other Aroclors estimated at zero.

This approach is considered conservative because highly elevated PCB concentrations are unlikely on the basis of samples collected from within the wastes in the western cell of the northern impoundments prior to initiation of the RI (TCEQ and USEPA 2006). In that study, Aroclors were never detected, even though Aroclor detection limits were much lower (<90 µg/kg). Elevated Aroclors are also considered unlikely based on results for several samples with normal Aroclor detection limits that were collected for the RI at the same time and even in the same core as those with interferences. For example, in SJGB011, Aroclor 1254 in the sample from 6 to 8 feet (182 to 243 cm) deep was not detected at a detection limit of 2,250 µg/kg, but in the same core, in the sample interval from 10 to 12 feet (304 to 350 cm) deep, Aroclor 1254 was not detected at a detection limit of 9.5 µg/kg. In summary, there is uncertainty about the actual Aroclor concentrations in the materials collected from within the 1966 perimeter of the northern impoundments. However, the absence of Aroclor detections in sediment or waste samples collected by TCEQ and USEPA (2006), and in other samples closely proximal to the samples that had matrix interferences, confirms that the approach taken to estimating total PCBs in sediment is conservative.

[14] The USEPA comment requiring evaluation of exposures to total PCBs as the sum of 43 specific congeners was first articulated in the comments on the Tissue SAP, which was produced after the Sediment SAP was final and implemented. See Appendix C of the Tissue SAP (Integral 2010b).

In the calculation of exposure point concentrations (EPCs) and in statistical evaluations of the datasets (e.g., characterization of data distributions), specific rules were applied for estimating values for censored data. Data distributions for each medium in each exposure unit were tested using the Shapiro-Wilk test for normality (Johnson et al. 2007). Procedures for substituting values for censored data varied, depending on the sample size and the detection frequency, as follows:

- For each dataset used in calculation of an EPC, the detection frequency was calculated as the percentage of values not flagged with a "U" qualifier (not detected).
- Nondetects in datasets with sample sizes equal to or greater than 10 and detection frequencies equal to or greater than 50 percent were set to one-half the detection limit and were included in all calculations.
- Datasets with sample sizes equal to or greater than 10 and detection frequencies between 20 and 50 percent were addressed using statistical substitution methods. The substitution method used depended on the distribution of the dataset; for normally or lognormally distributed data, upper confidence limits on the mean (UCLs) were estimated using robust regression on order statistics (Helsel 2005); for datasets with unknown data distributions (those that could not be defined as normal or lognormal), a nonparametric Kaplan-Meier approach for inputting nondetects was used (Helsel 2005; Singh et al. 2006).
- Nondetects in datasets with sample sizes less than 10, regardless of detection frequency, or in datasets with detection frequencies less than 20 percent, regardless of sample size, were not subject to statistically derived substitutions because the pool from which information about the data distribution could be drawn was insufficient for robust substitution methods. These datasets were treated with nondetects substituted at one-half the detection limit.

4 TOXICITY ASSESSMENT

The toxicity assessment summarizes the health effects that may be associated with exposure to the $COPC_H$s selected for the risk assessment and identifies doses that may be associated with those effects. Toxicological criteria are numerical expressions of dose and response and are used along with estimates of exposure to calculate potential risks to human receptors. These criteria may differ, depending on the duration and route of exposure. Therefore, the toxicological criteria required for this BHHRA were selected to reflect exposure routes represented in the CSMs. Toxicological criteria for cancer and noncancer effects are available.

The TESM (Integral 2012b; Appendix B) presents the cancer- and noncancer-based toxicological criteria that were used in this BHHRA for the $COPC_H$s identified for the area north of I-10 and aquatic environment, as well as for thallium.[15] At the time the TESM was prepared, sampling efforts for the area of investigation on the peninsula south of I-10 were ongoing and the complete set of $COPC_H$s for this area had not yet been developed. Additional $COPC_H$s for the area of investigation on the peninsula south of I-10 are identified in Appendix C and Appendix M and toxicological criteria for the additional $COPC_H$s are documented in Appendix D of this BHHRA. The cancer- and noncancer-based toxicological criteria selected for all $COPC_H$s are summarized in Tables 4-1 and 4-2, respectively.

This section describes the methods that were used for selecting toxicological criteria for the final $COPC_H$s, and provides a summary of the bases of the criteria selected. Because the toxicity of dioxin-like compounds (DLCs) is expressed in this BHHRA using TEQ values, a brief overview of the TEQ approach, which relates to the mechanism of action by which these compounds are believed to act and to the relative potency of the various DLCs, is also provided below.

[15] Thallium was not selected as a $COPC_H$ for the northern impoundments; however, the maximum concentration of thallium measured in the area of investigation in the peninsula south of I-10 during the Phase 1 2011 soil sampling event exceeded the industrial screening value. Although this maximum concentration was measured in a deep subsurface soil sample (i.e., 8-foot interval), thallium was addressed in the TESM in anticipation that it might be identified as a $COPC_H$ for the area of investigation in the peninsula south of I-10. Ultimately, thallium was not selected as a $COPC_H$, and, therefore, it is not discussed further in this toxicity evaluation.

4.1 Hierarchy for Selecting Toxicological Criteria

In accordance with procedures outlined by USEPA (2003a), the following hierarchy of sources was considered in selecting toxicological criteria for this BHHRA, in order of preference:

- Tier 1: USEPA's IRIS[16]
- Tier 2: USEPA's Provisional Peer Reviewed Toxicity Values from the National Center for Environmental Assessment/Superfund Health Risk Technical Support Center[17]
- Tier 3: Other USEPA and non-USEPA sources, such as the Agency for Toxic Substances and Disease Registry (ATSDR) minimal risk levels,[18] USEPA's Health Effects Assessment Summary Tables (HEAST; USEPA 1997b), California Environmental Protection Agency values,[19] and other sources that are current, publicly available, and have been peer reviewed.

4.2 Toxic Equivalency Factors for Dioxin-Like Compounds

In all, there are 75 dioxins and 135 furans that are differentiated by the numbers and positions of the chlorine atoms present. Seventeen of those congeners have chlorine substitutions in the 2,3,7,8- positions of the molecule. It is widely believed that toxicity of these 17 congeners occurs through a common biochemical mechanism, one that is initiated by the binding of the congener to the aryl hydrocarbon receptor (AhR), and leads to alterations in gene expression and signal transduction that are believed to be the biochemical determinants of toxic effects (Birnbaum 1994). Similarly, 12 coplanar PCB congeners have been shown to act via the same AhR mechanism and, therefore, are considered to be "dioxin-like." Of the 17 dioxin and furan congeners and 12 coplanar PCB congeners, TCDD has been the most extensively studied and exhibits the greatest potential for toxicity. Toxicological information on the other DLCs is more limited.

Because of the limited toxicological information for many of these DLCs, the TEQ approach was developed. Under the TEQ approach, the magnitude of toxicity of each of the dioxin-

[16] Available at: http://www.epa.gov/ncea/iris/.
[17] Values available at: http://hhpprtv.ornl.gov/
[18] Available at: http://www.atsdr.cdc.gov/mrls/index.asp
[19] Available at: http://www.oehha.ca.gov/air/hot_spots/tsd052909.html

like congeners is related to the toxicity of TCDD using a congener-specific toxic equivalency factor (TEF). The concentration of each congener is converted to an equivalent concentration of TCDD by multiplying the concentration of the congener by its TEF to derive a TEQ concentration for that congener. The congener-specific TEQs are then added together to compute the total TEQ concentration of the mixture of dioxins and furans (i.e., TEQ_{DF}) and of dioxin-like PCBs (i.e., TEQ_{P}). The resulting TEQ concentrations provide the metric to be used in evaluating exposure to the mixtures.

While there are substantial uncertainties associated with the use of TEQs (see Appendix B), USEPA generally requires that the TEQ approach be used to evaluate the risks due to mixtures of dioxins and furans. The TEQ approach therefore has been used in this BHHRA to estimate potential health effects associated with mixtures of dioxins and furans, and dioxin-like PCBs.

4.3 Cancer Effects

USEPA evaluates the potential for individual chemicals to cause cancer in humans. An initial step in this evaluation is a qualitative, weight-of-evidence (WOE) evaluation of the extent to which a chemical is believed to be a human carcinogen based on the results of human and/or animal studies. For those chemicals that have been categorized as known or probable carcinogens, USEPA typically develops chemical-specific cancer slope factors (CSFs), which are upper-bound estimates of the carcinogenic potency. These CSFs are used to estimate the incremental risk of developing cancer, corresponding to a lifetime of exposure at the levels described in the exposure assessment. Under USEPA's standard default risk assessment procedures, estimates of carcinogenic potency reflect the conservative assumption that there is no threshold dose for carcinogenic effects; that is, there is no entirely "safe" dose and exposure to any amount of the chemical will contribute to an individual's overall risk of developing cancer during a lifetime.

USEPA's *Guidelines for Carcinogen Risk Assessment* (2005), however, recognizes that some carcinogens act in a manner within the body (i.e., a mode of action) that follows a nonlinear, threshold response, similar to the threshold dose assumed when developing toxicological criteria for noncancer effects. A nonlinear dose-response relationship is one in which a level of exposure exists at which there is no increased risk of cancer within the exposed population so that only exposure levels that exceed the threshold dose will result in an increased probability of developing cancer. USEPA allows for estimates of carcinogenic potency to be based on a non-linear model when sufficient evidence exists to support a non-linear mode of action for the general population and any subpopulations of concern (USEPA 2005).

4.3.1 Dioxins and Furans

No Tier 1 or Tier 2 criterion is available to evaluate the potential carcinogenic effects of TCDD and other DLCs. Therefore, it was necessary to consider Tier 3 sources in selecting a cancer-based criterion for use in this BHHRA.

USEPA has been conducting an assessment of dioxin risks (the "dioxin reassessment") for nearly 20 years, but this process is not yet complete. During this period, there has been extensive, worldwide evaluation of the toxicological literature for dioxin and furans, and substantial disagreement remains within the scientific community as to the appropriate approach for estimating the toxicological potential of these compounds. Available Tier 3 values vary widely in both magnitude and approach, as discussed in Appendix B.

The available Tier 3 values for the carcinogenic potential of TCDD can be broken into two categories. The first category includes those criteria that are based on the assumption that a CSF for TCDD should be derived using a linear dose response model. The second category includes those toxicological criteria that are based on the assumption that there is a threshold dose for TCDD's carcinogenic activity so that this threshold must be reached before TCDD can exert a carcinogenic effect.

USEPA has historically used a linear dose response model to evaluate the potency of TCDD and other DLCs. There is, however, a growing consensus worldwide, including among members of USEPA's Science Advisory Board (SAB) and the National Academies of Sciences, that there is likely a threshold for TCDD's carcinogenicity and that it should be evaluated using a nonlinear, threshold approach (WHO 1998; JECFA 2002; Simon et al. 2009; NAS 2006; ACC 2010; TCEQ 2010a,b, 2011; Haney 2010).

For this BHHRA, a threshold based tolerable daily intake (TDI) of 2.3 pg/kg-day was used to evaluate potential cancer effects resulting from assumed exposure to dioxins and furans. The Joint Food and Agriculture Organization/World Health Organization Expert Committee on Food Additives (JECFA) derived a threshold-based toxicity criterion for TCDD based on body burden rather than on administered dose. This committee included individuals from the U.S. Food and Drug Administration (U.S.), Health Canada (Canada), the National Institute of Public Health and the Environment (Netherlands), Municipal Institute of Medical Research (Spain), Chemisches and Veterinäruntersuchungsamt (Germany), Scientific Directorate on Human Nutrition and Food Safety of the National Institute for Agricultural Research (France), Center for Risk Management (U.S.), and the National Institute of Public Health and the Environment (Netherlands). These individuals reviewed all of the available scientific literature related to the toxicology of dioxins and furans in both animals and humans that was available at that time. Based on their comprehensive review and analysis, the committee concluded that there was a threshold for all toxic effects associated with exposure to TCDD, including cancer, and that developmental effects represented the most sensitive of all of the toxic endpoints. They concluded that a toxicological criterion based on noncancer effects would also address any potential cancer risk. This conclusion was supported by the subsequent studies conducted by Simon et al. (2009) and NTP (2006).

JECFA concluded that the tolerable monthly intake of 70 pg/kg-month (equivalent to a TDI of 2.3 pg/kg-day) was a reliable value from animal studies that could be used to assess both cancer and noncancer effects of dioxin. Because this value was developed by an expert panel, USEPA (2010b) considers it to be adequately peer reviewed so that it represents a Tier 3 value. This value is well-supported by the toxicological literature and an international panel of scientists, and is consistent with SAB comments on the dioxin reassessment and the opinions of other toxicologists who support the use of a threshold approach in developing toxicological criteria for DLCs (NAS 2006; Simon et al. 2009; TCEQ 2009, 2010a, 2011). This value was used to evaluate the potential carcinogenic effects of TEQ_{DF} in this BHHRA.

Alternative Tier 3 criteria derived from linear dose response models were presented and discussed in the TESM (Appendix B). These were used for calculating cancer risks that are presented and discussed as part of the uncertainty evaluation.

4.3.2 PCBs

PCBs are a large family of 209 related congeners. These compounds range from mono-chlorinated congeners (having only one chlorine atom) to fully substituted deca-chlorinated congeners (with chlorine at all possible ring locations). Most of the PCBs that are found in the environment were released as commercial mixtures that were originally sold in the U.S. under the trade name Aroclor. Generally, Aroclors were identified by trade names such as Aroclor 1254.

According to USEPA, the cancer potency of PCB mixtures depends on the media of interest and the PCB congeners present. USEPA's Integrated Risk Information System (IRIS) database provides an upper bound CSF of 2 $(mg/kg\text{-}day)^{-1}$ and central tendency CSF of 1 $(mg/kg\text{-}day)^{-1}$ for PCB mixtures. These CSFs were used to estimate upper-bound and central tendency cancer risks, respectively, associated with total PCBs (either sum of 43 congeners[20] or sum of Aroclors).

In addition, TEFs have been developed for the 12 PCB congeners that are assumed to be DLCs because they also have a high affinity to bind to the AhR. Therefore, for the uncertainty analysis, an equivalent concentration of TCDD for the PCB mixture (i.e., TEQ_P) was evaluated using the toxicological criterion for TCDD.

4.3.3 Other COPCs

IRIS provides CSFs of 0.014 $(mg/kg\text{-}day)^{-1}$ for bis(2-ethylhexyl)phthalate and 1.4 $(mg/kg\text{-}day)^{-1}$ for inorganic arsenic (Table 4-1). These values were used to evaluate the potential carcinogenic risks due to these $COPC_{HS}$. The bases of these values are provided in the TESM (Appendix B).

In addition to a subset of the COPCHs already identified for the area north of I-10 and aquatic environment, benzo(a)pyrene was identified as a $COPC_H$ for the area of investigation south of I-10. IRIS provides a CSF for benzo(a)pyrene of 7.3 $(mg/kg\text{-}day)^{-1}$. The basis of this value is provided in Appendix D, *Supplemental Toxicological and Chemical-Specific Parameters.*

[20] Total PCB concentrations were calculated as the sum of the 43 congeners shown in Table 3-3.

All other $COPC_{HS}$ are not considered to have carcinogenic potential via oral exposure routes and, therefore, were not included in the estimation of potential cancer risks.

4.4 Noncancer Effects

For chemicals that are considered to have the potential to cause noncancer health effects, toxicological criteria are based on the adverse health effect elicited at the lowest doses evaluated in animal or human studies. The dose level at which no adverse effects are observed (i.e., the no-observed-adverse-effect level [NOAEL]), or the lowest dose tested at which adverse effects are observed (i.e., the lowest-observed-adverse-effect level [LOAEL]), is the point of departure (POD) for developing noncancer toxicological criteria. Uncertainty and/or modifying factors are typically applied to the POD to adjust for uncertainties in the toxicity data, differences in responses among animal species and humans, and variations in inter-individual sensitivity within the human populations. This provides a margin of safety to ensure that the estimated dose level selected as the criterion will not result in adverse health effects in the exposed human population. The resulting toxicological criterion, known as the reference dose (RfD), is the dose level at or below which no adverse health effects are expected to occur.

To evaluate potential noncancer health effects that may result from exposure to a chemical, the potential hazard is evaluated by comparing the estimated daily intake with an RfD. RfDs are available for different durations of exposure. For long-term exposures, this is identified as a chronic RfD. USEPA (1989) defines the chronic RfD as a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime. Subchronic RfDs are used to evaluate potential noncancer hazards associated with exposures of less than 7 years.

4.4.1 Dioxins and Furans

USEPA's IRIS database provides an RfD of 0.7 pg/kg-day for TCDD based on developmental effects reported by two epidemiological studies (Table 4-2). This criterion was used to evaluate the potential noncancer hazards associated with TEQ_{DF}.

4.4.2 PCBs

USEPA's IRIS database provides an RfD of 2×10^{-5} mg/kg-day for Aroclor 1254-based changes in immune response measured in rhesus monkeys dosed with Aroclor 1254 compared to controls. This criterion was used to evaluate potential noncancer hazards due to exposures to total PCBs (i.e., sum of 43-congeners or sum of Aroclors) in Site-related media.

IRIS does not discuss the approach to be used for evaluating noncancer effects of dioxin-like PCB congeners and USEPA has not yet made any policy statements about the adoption of the RfD for TCDD for PCB risk assessment. In addition, there is no indication that the endpoints that were selected as the basis for the TCDD RfD are also associated with PCB toxicity. This means that the application of the TCDD RfD to dioxin-like PCBs is likely to result in substantial uncertainty in estimates of the risks due to PCBs. However, in the event that USEPA may require that the TEQ approach also be used to evaluate noncancer effects of total TEQ mixtures, an evaluation of noncancer hazards using this approach was completed and discussed in the uncertainty analysis.

4.4.3 Other COPCs

IRIS provides chronic RfDs for the remainder of the $COPC_{H}s$ for the area north of I-10 and the aquatic environment and the area of investigation south of I-10 with the exception of organic forms of arsenic and copper. The chronic RfDs for organic arsenic and copper were taken from ATSDR and HEAST, respectively (Table 4-2). These RfDs were used for evaluating potential chronic exposures to these $COPC_{H}s$. The critical endpoint for each $COPC_{H}$ is also provided. The specific bases of these values are provided in the TESM (Appendix B).

4.4.4 Subchronic Noncancer Effects

Subchronic RfDs are used to evaluate potential noncancer hazards associated with exposures between 2 weeks and 7 years (USEPA 1989). The trespasser scenario for the area of investigation south of I-10 represents the only scenario with exposure durations in this range and where subchronic exposures are therefore relevant. Although there is generally adequate information on toxicological criteria to evaluate long-term or chronic exposures, information on subchronic exposures is more limited. No subchronic RfDs are available for any of the

$COPC_{H}s$ identified for the noncancer evaluation (i.e., for dioxins and furans and for inorganic arsenic) for soils 0 to 12 inches deep; therefore, the chronic RfDs were used to evaluate potential noncancer hazards associated with the hypothetical trespasser scenario (Table 4-2). The subchronic RfD for PCBs (i.e., a $COPC_{H}$ for soils 0 to 10 feet deep) was used to evaluate potential noncancer hazards for hypothetical future construction worker scenarios (Table 4-2). As discussed in Appendix D, no published subchronic RfD is available for benzo(a)pyrene. Therefore, this chemical was evaluated for its carcinogenic potential only.

5 EXPOSURE AND RISK CHARACTERIZATION FOR AREA NORTH OF I-10 AND AQUATIC ENVIRONMENT

This section presents the exposure assessment and risk characterization for the area north of I-10 and the aquatic environment. The purpose of the exposure assessment (Section 5.1) is to estimate the type and magnitude of potential human exposure to COPCs identified at a site. In the risk characterization (Section 5.2), these estimates of exposure are combined with toxicological criteria to yield numerical estimates of potential adverse health effects to humans.

5.1 Exposure Assessment

For this BHHRA, potential exposures under the baseline condition (i.e., immediately prior to the TCRA) were first estimated using deterministic methods. The exposure scenarios, algorithms, and assumptions used for the deterministic assessment were established and discussed in the EAM (Appendix A) and are summarized below. For risk assessment purposes, the baseline levels of exposure are assumed to apply throughout the exposure duration for each hypothetical scenario, even though there is no basis for assuming that baseline represents conditions that existed at any other point in time, or that would have continued to exist in the absence of the TCRA.

This set of assumptions was also used for estimating background and post-TCRA exposures for those scenarios that were selected for further analysis (i.e., see Figure 1-4). For any scenario selected for further analyses, potential exposures for each component exposure pathway were additionally estimated using probabilistic methods. The inputs for probabilistic analysis are briefly discussed below and are presented in detail in Appendix G.

5.1.1 Exposure Scenarios

Three potential receptor groups were assumed for the quantitative risk assessment for the area north of I-10 and the aquatic environment: a hypothetical recreational fisher, a hypothetical subsistence fisher, and a hypothetical recreational visitor. Based on the CSM for the area north of I-10 and aquatic environment, the following potential exposures were quantified for these hypothetical receptor groups:

- Recreational Fisher—direct contact (incidental ingestion and dermal contact) with sediment and soils, ingestion of finfish, and ingestion of shellfish
- Subsistence Fisher—direct contact (incidental ingestion and dermal contact) with sediment and soils, ingestion of finfish, and ingestion of shellfish
- Recreational Visitor—direct contact (incidental ingestion and dermal contact) with sediment and soils.

Both hypothetical recreational and subsistence fishers are assumed to ingest fish and/or shellfish caught within USEPA's Preliminary Site Perimeter. Detailed information regarding fishing activities and consumption patterns in this area is not available. In the absence of this specific information on consumption of fish from the area, exposures were estimated separately under three general scenarios: 1) finfish ingestion only, 2) clam ingestion only, and 3) crab ingestion only. Focusing the risk assessment on single-tissue type exposures is conservative because it identifies and quantifies potential exposure to the tissue type that result in the highest potential for exposure. In estimating cumulative exposure, estimated exposures from the direct contact pathways (i.e., ingestion and dermal contact) were summed with exposures for each tissue ingestion scenario separately.

A series of hypothetical exposure scenarios were considered for each receptor based on tissue type ingested as well as the exposure units defined for sediments. The exposure units identified, and resulting scenarios evaluated for this risk assessment are described below.

5.1.1.1 Exposure Units

An exposure unit is defined as the area within which the receptor group being evaluated is expected to move and encounter environmental media for the duration of the exposure (USEPA 2002a). Selection of exposure units should also consider the statistical characteristics of the datasets (USEPA 2002a) where concentrations of COPCs in environmental media vary spatially; exposure units are selected to allow the risk assessment to distinguish between those areas of a site that present higher potential for risk to the exposed population and those areas that present lower potential risks. Such a distinction can facilitate risk management decisions by indicating which areas are associated with the highest risk, and therefore, which areas should be prioritized for risk reduction. Exposure units for this BHHRA were identified by following the DQOs established in the RI/FS Work Plan (Anchor QEA and Integral, 2010) and in the Tissue SAP (Integral 2010b). The process used to define exposure units and the results of that analysis are documented in detail in the EAM (Appendix A). Figures 5-1 through 5-3 show the exposure units identified for baseline sediments, tissue, and soils respectively. Nearshore sediment samples were collected as part of the RI from five beach areas within USEPA's Preliminary Site Perimeter. A statistical analysis of the available data indicated that, except for Beach Areas B and C, the sediment concentrations in these areas were sufficiently different that they should not be combined (Figure 5-1) (Appendix A, Section 3.4). Three FCAs were identified at the Site (Figure 5-2). Statistical analysis of the fish tissue data indicated that FCAs 2 and 3 could be combined for catfish fillets and crabs, and FCAs 1 and 3 could be combined for clams (Appendix A, Section 3.4). For soils a single exposure unit was defined. Figure 5-3 shows the locations of the samples used to define this exposure unit. The selection of a single exposure unit for soils north of I-10 was based on the assumption that individuals visiting the area north of I-10 could have direct contact with soils in all of the sample collection areas during their visit.

Based on the analysis summarized above, the following exposure units were defined for the baseline condition:

- Sediments
  - Beach Area A
  - Beach Area B/C—consisting of data pooled from Beach Areas B and C
  - Beach Area D
  - Beach Area E
- Hardhead catfish fillet
  - FCA 2/3—consisting of data pooled from FCA 2 and FCA 3
  - FCA 1
- Edible crab
  - FCA 2/3—consisting of data pooled from FCA 2 and FCA 3
  - FCA 1

- Edible clam
  - FCA 1/3—consisting of data pooled from FCA 1 and FCA 3
  - FCA 2
- Soils
  - The entire area north of I-10

Fencing constructed as part of the TCRA now limits regular access to all Beach Areas except Beach Area A. Therefore, Beach Area A was defined as the only exposure unit for sediments under the post-TCRA condition. There is future potential for receptors to access Beach Areas B and C/D (e.g., in the case that a breach in the fencing was to occur). The impact of such access on potential exposure and associated risk under the post-TCRA condition is described in the uncertainty evaluation for this BHHRA. In addition, given this more limited access, a smaller area was considered as the post-TCRA exposure unit for soils. Figures 5-4 and 5-5 show the post-TCRA exposure units for sediments and soils, respectively. The exposure units assigned for post-TCRA tissue remain unchanged from baseline.

5.1.1.2 Resulting Hypothetical Exposure Scenarios

Exposure units for various media were combined to represent exposures that could hypothetically occur under the assumed conditions. For instance, hypothetical fishers at Beach Area A are assumed to have direct contact with sediments at Beach Area A, and to catch and ingest finfish from FCA 2/3, crabs from FCA 2/3, or clams from FCA 1/3. Hypothetical fishers at Beach Area D are assumed to have direct contact with sediments in Beach Area D and assumed to catch and ingest finfish from FCA 1, crabs from FCA 1, or clams from FCA 1/3. The complete set of hypothetical exposure scenarios evaluated for the baseline condition for this BHHRA is provided in Table 5-1.

5.1.2 Estimates of Exposure

This section presents the equations and exposure parameters that were used for estimating exposure for this BHHRA. USEPA (1993) guidance recommends that two types of exposure estimates be calculated. The reasonable maximum exposure (RME) is defined as the highest exposure that could reasonably be expected to occur for a given exposure pathway and scenario at a site. The RME is intended to account for uncertainty in the chemical concentration at the point of exposure, and for variability and uncertainty in exposure parameters. USEPA also recommends that the central tendency exposure (CTE), or average estimate of exposure, be presented in a risk assessment. Both RME and CTE estimates were calculated for this BHHRA. In addition, for any exposure scenario that was selected for further evaluation (Figure 1-4), a PRA was employed to estimate exposure. The equations and exposure parameters used in the risk assessment are presented below.

5.1.2.1 Equations

Three types of potential exposures were evaluated: 1) ingestion of sediment and/or soil, 2) dermal absorption of sediment and/or soil, and 3) ingestion of fish and/or shellfish. The equations that were used to calculate these potential exposures are presented below. The equations are common to both the deterministic and probabilistic evaluations.

Equation 5-1. Intake via Ingestion of Soil and/or Sediment

Relevant Receptor Groups: fishers, recreational visitors

$$I_{soil-sed} = \frac{\left[\left(C_{soil} \times IR_{soil} \times F_{soil}\right) + \left(C_{sed} \times IR_{sed} \times F_{sed}\right)\right] \times RBA_{soil-sed} \times FI_{soil-sed} \times EF_{soil-sed} \times ED \times CF_1}{BW \times AT} \quad \text{(Eq. 5-1)}$$

Where:

$I_{soil-sed}$	=	intake, the mass of a chemical contacted in soil and sediment by the receptor per unit body weight per unit time (mg/kg-day)
C_{soil}	=	chemical concentration in soil contacted over the exposure period (i.e., EPC for soil) (mg/kg)
IR_{soil}	=	soil ingestion rate (mg/day)
F_{soil}	=	fraction of total ingestion that is soil (percent as a fraction)
C_{sed}	=	chemical concentration in sediment contacted over the exposure period (i.e., EPC for sediment) (mg/kg)
IR_{sed}	=	sediment ingestion rate (mg/day)
F_{sed}	=	fraction of total ingestion that is sediment (percent as a fraction)
$RBA_{soil-sed}$	=	relative bioavailability adjustment for soil and sediment (percent as a fraction)

$FI_{soil\text{-}sed}$	=	fraction of total daily soil/sediment intake that is site-related (percent as a fraction)
$EF_{soil\text{-}sed}$	=	exposure frequency (days/year)
ED	=	exposure duration (years)
CF_1	=	conversion factor (1×10^{-6} kg/mg)
BW	=	body weight (kg)
AT	=	averaging time (days)

Equations 5-2 and 5-3. Dermal Absorbed Dose via Contact with Soil and Sediment

Relevant Receptor Groups: fishers, recreational visitors

$$DAD_{soil-sed} = \frac{DA_{event} \times SA \times EF_{soil-sed} \times FI_{soil-sed} \times ED \times EV}{BW \times AT} \qquad \text{(Eq. 5-2)}$$

Where:

$DAD_{soil\text{-}sed}$	=	dermal absorbed dose from soil and sediment (mg/kg-day)
DA_{event}	=	absorbed dose per event (mg/cm^2)
SA	=	skin surface area available for contact (cm^2)
EV	=	event frequency (day^{-1})

And

$$DA_{event} = \left[\left(C_{soil} \times AF_{soil} \times F_{soil}\right) + \left(C_{sed} \times AF_{sed} \times F_{sed}\right)\right] \times ABS_d \times CF_1 \qquad \text{(Eq. 5-3)}$$

Where:

AF_{soil}	=	adherence factor for soil (mg/cm^2)
AF_{sed}	=	adherence factor for sediment (mg/cm^2)
ABS_d	=	dermal absorption factor for soil/sediment (percent as a fraction)

Equation 5-4. Intake via Ingestion of Fish and Shellfish

Relevant Receptor Groups: fishers

$$I_{tissue} = \frac{C_{tissue} \times (1 - LOSS) \times IR_{tissue} \times RBA_{tissue} \times FI_{tissue} \times EF_{tissue} \times ED \times CF_2}{BW \times AT} \qquad \text{(Eq. 5-4)}$$[21]

Where:

I_{tissue}	=	intake, the mass of a chemical contacted in fish or shellfish tissue by the receptor per unit body weight per unit time (mg/kg-day)
C_{tissue}	=	chemical concentration in fish or shellfish tissue contacted over the exposure period (i.e., EPC for fish or shellfish) (mg/kg)
LOSS	=	chemical reduction due to preparation and cooking (percent as a fraction)
IR_{tissue}	=	fish or shellfish ingestion rate (g/day)
RBA_{tissue}	=	relative bioavailability adjustment for tissue (percent as a fraction)
FI_{tissue}	=	fraction of total fish or shellfish intake that is site-related (percent as a fraction).
EF_{tissue}	=	exposure frequency for fish or shellfish consumption (days/year)
CF_2	=	conversion factor (1×10^{-3} kg/g)

5.1.2.2 Deterministic Exposure Evaluation

The EPCs and exposure parameters selected for each scenario are summarized below and are discussed in detail in the EAM (Appendix A).

5.1.2.2.1 Exposure Point Concentrations

EPCs were estimated for each medium in each exposure unit according to the procedures outlined in Section 3.2. Tables 5-2 through 5-4 summarize the RME and CTE EPCs used for the deterministic assessment of baseline risks. Table 5-5 shows the EPCs for the deterministic assessment of background risks. Supporting documentation for the EPC derivations, including summaries of the best-fit distribution and basic summary statistics for each dataset, is provided as Appendix E.

[21] The equation presented here uses the term tissue generically to present parameters for finfish and shellfish. Intake of finfish and shellfish were estimated separately.

Post-TCRA risks were evaluated for dioxins and furans only. Data or representative concentrations for all $COPC_{HS}$ in all media of interest for post-TCRA conditions were not available and, therefore, dioxins and furans were used to provide a relative measure of hazard and/or risk. EPCs representative of post-TCRA conditions for each medium were estimated using a variety of methods. For sediments and soils, the portion of the baseline data from within the exposure units defined for the post-TCRA condition (i.e., defined as the areas that were still accessible to individuals following the TCRA) were used. No tissue data were collected following the TCRA. In the absence of such data, post-TCRA tissue concentrations for hardhead catfish were estimated using statistical relationships between baseline sediment and tissue samples established in the *Technical Memorandum on Bioaccumulation Modeling* (Integral 2010c). For clams and crabs, where no meaningful model for predicting sediment–tissue relationships existed, assumptions regarding the baseline dataset were used to estimate post-TCRA EPCs. Appendix F documents the detailed methods used for post-TCRA EPCs as well as the post-TCRA risk characterization results and the uncertainties associated with these estimates.

5.1.2.2.2 Exposure Parameters

This section provides an overview of the exposure assumptions used in the deterministic evaluation. A detailed presentation and the supporting rationales for these assumptions are included in the EAM (Appendix A). A summary of these exposure parameters is presented in Table 5-6. Assumptions adopted for chemical specific exposure parameters are provided in Table 5-7.

Differences in activity and intake parameters have been characterized for younger children, older children, and adults. Therefore, exposure parameters were developed separately for young children (ages 1 to <7 years), older children (ages 7 to <18 years), and adults (ages 18 years and older).

Considering the exposure factors assumed for this BHHRA, young children would have higher potential exposures (on a per unit body weight basis) relative to other age groups. Therefore, for the RME scenarios for all human receptor groups evaluated, it was assumed that a portion of the total exposure occurs at these younger life stages. This is a conservative assumption because it results in an upper-bound RME scenario in which the calculated exposure for any alternative age grouping over the same chronic exposure duration would be lower. As established in the EAM, the individuals considered most likely to use the area under study under baseline conditions are adults. Therefore, for the CTE analysis, only adult exposures are evaluated. It is however, recognized that children may frequent the area along with adults. At the request of USEPA (comment 9 of the draft BHHRA, Appendix N) an additional CTE analysis for a hypothetical young child receptor was performed and is presented in the uncertainty evaluation.

Common Parameters

Given the lack of specific information on fishing and recreational behaviors within USEPA's Preliminary Site Perimeter, the exposure durations were conservatively based upon standard default assumptions used for residents. Default exposure durations of 33 years for the RME and 12 years for the CTE (USEPA 2011a) were based on studies of occupational mobility, and were adopted for this BHHRA.

Following common practice for human health risk assessment, the averaging time selected depended on the toxic endpoint (cancer or noncancer) being assessed. For noncarcinogens, the averaging time was set equal to the exposure duration (e.g., for an exposure duration of 6 years the averaging time was 2,190 days). For carcinogens that were evaluated with a CSF, the averaging time was set equal to a lifetime (i.e., 78 years or 28,470 days) (USEPA 1989, 2011a). When the toxicity of a carcinogen was described using a criterion that assumed a threshold dose was required for an adverse effect to be elicited (i.e., TEQ_{DF}) the averaging time was set equal to the exposure duration. This latter approach described for threshold based carcinogens is essentially the same as the approach used for evaluating noncancer endpoints.

For the deterministic evaluation, mean body weights of 19, 50, and 80 kg were selected for the young child, older child, and adult age groups, respectively. These body weights were based on data collected from the 1999–2006, National Health and Nutrition Examination Survey (NHANES), and recommended in USEPA's Exposure Factors Handbook (2011a).

Parameters for Tissue Ingestion

Assumed fish and shellfish ingestion rates were selected from a study of fishing activity and consumption conducted in Lavaca Bay, Texas (Alcoa 1998). Lavaca Bay, which covers roughly 40,000 acres, is part of the larger Matagorda Bay system. This system is similar in size to Galveston Bay and is situated further south along the Texas coastline. The demographics in the counties surrounding the two bays are similar (2010 Census data for Calhoun, Chambers, Galveston, Harris, Jackson, and Victoria counties).[22]

The Lavaca Bay study collected data about consumption rates, fraction ingested from a contaminated source area, and the species composition of the fish consumed. The study was conducted during the month of November, which was reported to be the month of highest fishing activity in the bay (Alcoa 1998) and nearly 2,000 anglers participated in the study. It was conducted for the specific use of supporting a risk assessment for the Alcoa Point Comfort/Lavaca Bay Superfund Site.

Lavaca Bay ingestion rates reported by Alcoa (1998) for finfish and shellfish were adopted for this BHHRA. They were selected because they are Texas-specific and represent consumption from a fishery that is similar to the fishery associated with the area inside USEPA's Preliminary Site Perimeter. For the hypothetical recreational fisher, mean rates were used for the CTE analysis, while the 95UCL rates were used for the RME analysis. Although the Lavaca Bay study did not identify a true subsistence population for that area, the study did present upper bound (90^{th} or 95^{th} percentile) estimates of ingestion rates for the surveyed groups. These rates were selected as RME ingestion rates for the hypothetical subsistence fisher. For each of these, the average of rates for men and women were assumed for the adult ingestion rates. The rates provided for youths in the study were used to evaluate the older child while the rates provided for small children were used to evaluate exposures to the young child. The exposure frequency for ingestion of tissue was assumed to be 365 days/year for all hypothetical fishers since the fish ingestion rates used were annualized average daily averages.

[22] http://factfinder2.census.gov/faces/nav/jsf/pages/index.xhtml

Given the relatively small spatial extent of the area within USEPA's Preliminary Site Perimeter compared to the size of the Galveston Bay fishery, it is unlikely that 100 percent of the fish consumed over the 33-year-exposure duration assumed for the RME would be harvested from the area of study. The survey conducted by Alcoa (1998) at Lavaca Bay segregated the consumption data by the areas fished; specifically, a 1,500-acre subarea (indicated as the closure area), other portions of Lavaca Bay, and areas outside of Lavaca Bay. Similar to conditions at Lavaca Bay, the waters associated with USEPA's Preliminary Site Perimeter represent a very small fraction of the Galveston Bay fishery. Also like Lavaca Bay, there are many other locations around Galveston Bay that can be used for fishing. Therefore, the data from the Lavaca Bay survey were informative for the purposes of this BHHRA.

It was assumed that 25 percent of the total fish consumed by RME hypothetical recreational fishers, and 10 percent of total fish consumed by CTE hypothetical recreational fishers were collected from within USEPA's Preliminary Site Perimeter. These values were applied for the fractional intake term (FI_{tissue}) for hypothetical recreational fishers in Equation 5-4, above. Their selection is conservative for this BHHRA, as less than one percent of the fish and shellfish consumed in Lavaca Bay was from the 1,500 acre sub-area being evaluated. A full discussion of the findings of the study is found in the EAM (Appendix A).

There was no information specific to the area within USEPA's Preliminary Site Perimeter available with which to estimate the fraction intake term (FI_{tissue}) in Equation 5-4, above, for the hypothetical subsistence fisher. If subsistence activities did occur in this area, it is possible that fishers participating in these activities could fish exclusively from the waters adjacent to the area. Given the lack of information specific to fishing behaviors in the area of study, a conservative fractional intake of 1.0 was adopted for the subsistence fisher scenario.

Parameters for Direct Contact

The majority of activity by a fisher was expected and assumed to occur along the water's edge so that substantial exposure to soil was not likely. Therefore, for the fishing scenarios, the fraction of total intake that was attributed to such soils was assumed to be zero, while the fraction of total daily intake from sediment was assumed to be 1.0 (100 percent). It was envisioned, however, that the recreational visitor who is not fishing might spend equal amounts of time in contact with soils and sediments. Therefore, the fraction of total exposures attributed to soils and sediments were both assumed to be 0.5 (50 percent).

Based on USEPA's (2011a) recommended ingestion rates for soil, soil and sediment ingestion rates of 20 mg/day were assumed for adults and used to evaluate both CTE and RME estimates. An ingestion rate of 50 mg/day was assumed for older children. For younger children, a rate of 125 mg/day was assumed.[23]

For the skin surface area parameter, surface areas of 6,080 and 4,270 cm^2 were assumed for the older child and adult, respectively (USEPA 2011a), based on the assumption that an individual's hands, forearms, lower legs, and feet may come into contact with soil and/or sediment. For young children playing in the soil and/or sediment, it was assumed that the entire surface area of the leg might be in contact with sediments in addition to the hands, forearms, and feet. Based on this assumption for the young child, a surface area of 3,280 cm^2 was used (USEPA 2011a). The same surface areas were used to evaluate both the CTE and RME conditions.

Following USEPA recommendations, weighted adherence factors were calculated for each age group. These were based on the surface areas of the assumed, exposed body parts and body-part-specific adherence factors presented by USEPA (2011a) that were based on studies completed in sediment, and soil.

For sediment exposure estimates, weighted adherence factors of 3.6, 5.1, and 4.9 mg/cm^2 for young children, older children, and adults, respectively, were derived based on a study of children playing in sediment. The study was recommended by USEPA (2011a) and was one of the only available studies that investigated sediment adherence to skin. Given the difference in sediment types within USEPA's Preliminary Site Perimeter compared to those present in the study used to develop the factors presented in USEPA (2011a), and the importance of sediment type in predicting soil adherence (Spalt et al. 2009), uncertainty was

---

[23] Rates for the older child and young child are for the RME scenario. No child component was considered in the CTE scenario for the hypothetical recreational fisher and visitor. No CTE evaluation was completed for the hypothetical subsistence fisher scenarios.

introduced in the exposure estimates by the use of this factor. This uncertainty is further discussed within the uncertainty evaluation of the risk characterization.

A weighted soil adherence factor of 0.07 mg/cm^2 was calculated for older children and adults using data that described the adherence of soils to skin in adults participating in a variety of activities (USEPA 2011a). Data from a study conducted in children exposed to soil were used to derive a soil adherence factor of 0.09 mg/cm^2 for young children (USEPA 2011a).

The assumed exposure frequency for the direct contact pathways was based on estimates of the number of trips to the area within USEPA's Preliminary Site Perimeter each year. According to the 2006 survey of Texas anglers conducted by the U.S. Fish and Wildlife Service (USFWS), the mean number of days spent fishing marine waters by Texas residents was 13 days/year (USFWS 2006). This value was assumed for the CTE exposure frequency for direct contact pathways for the hypothetical recreational fisher. It is reasonable to assume that more avid anglers may fish with a higher frequency than the average. A survey of Maine's freshwater anglers (Ebert et al. 1993), found that the 95^{th} percentile frequency of fishing trips per year was nearly three times that of the average number of fishing trips per year. Based on this relationship, an RME frequency of 39 days/year was assumed for the hypothetical recreational fisher. It is reasonably anticipated that hypothetical subsistence fishers, if present, may participate in fishing activities more often than recreational fishers; however, it is not likely that they would fish the same location more than an average of 2 days per week, every week of the year, over the entire assumed exposure duration of 33 years. Thus, an RME exposure frequency for direct contact pathways of 104 days/year was assumed for the hypothetical subsistence fisher scenario.

In the absence of data concerning recreational use of the area within USEPA's Preliminary Site Perimeter, RME and CTE frequencies of 104 and 52 days per year, respectively, were assumed for hypothetical recreational visitors. These were based on assumed average frequencies of 2 days per week and 1 day per week throughout the course of the year, respectively.

It is not anticipated that a fisher's or a visitor's direct contact with soils and/or sediments would typically be limited to the area within USEPA's Preliminary Site Perimeter. These individuals would likely not spend the entire day on each day that they fish or visit within this area; rather they might spend only a few hours and spend the remainder of those days engaged in activities in other areas where they could be exposed to soils or sediment from areas outside of USEPA's Preliminary Site Perimeter. No information specific to the area of study is available with which to estimate the fractional intake term for soil/sediment ($FI_{soil\text{-}sed}$) in Equation 5-1, above. Based on best professional judgment, a conservative fractional intake of 1.0 was adopted for the RME hypothetical recreational fisher and recreational visitor scenarios, and for the hypothetical subsistence fisher scenario. A fractional intake of 0.5 was adopted for the CTE scenario evaluated for the hypothetical recreational fisher and recreational visitor populations.

Chemical-Specific Factors

In addition to the scenario-specific exposure assumptions described above, there are a number of chemical-specific factors that were required to estimate $COPC_H$-specific exposure levels. These included oral bioavailability factors, dermal absorption factors, and reductions in chemical concentrations of certain $COPC_H$s due to preparation and cooking. The chemical-specific values used are summarized in Table 5-7 and are briefly discussed below. A more comprehensive discussion of these parameters and the rationales for the values selected were included in the EAM (Appendix A).

Relative Oral Bioavailability

Bioavailability refers to the degree to which a substance becomes available to the target tissue after administration or exposure (USEPA 2012c). Relative bioavailability is a measure of the extent of absorption that occurs for different forms of the same chemical, different dosing vehicles, or different dose levels. Relative bioavailability adjustment (RBA) factors for oral pathways are used to account for the differences in chemical bioavailability in specific exposure media (i.e., soil, sediment, tissue) compared to the dosing vehicle used in the critical toxicity study that provides the basis for the $COPC_H$-specific toxicity criteria selected for use in this BHHRA.

The RBA can be expressed as:

$$RBA = \frac{absorbed\ \ fraction\ \ from\ \ exposure\ \ medium\ \ on\ site}{absorbe\ d\ fraction\ \ from\ \ dosing\ \ medium\ \ used\ in\ toxicity\ \ study} \qquad \text{(Eq. 5-5)}$$

In the absence of data from peer-reviewed publications or site-specific data on bioavailability of chemicals in sediment, USEPA and the Interstate and Technology Regulatory Council recommend that default factors for soil be adopted to evaluate sediment exposures (USEPA 2004; ITRC 2011). Sufficient data to determine $RBA_{soil\text{-}sediment}$ were available for dioxins and furans and for arsenic and these are discussed below. These chemical-specific RBAs were applied to the calculation of exposures via incidental ingestion of soil and sediment.

An $RBA_{soil\text{-}sediment}$ of 0.50 was adopted for dioxins and furans. This value was derived from data on the bioavailability of TCDD in soils from a range of studies selected and presented by USEPA (2010d) in their *Final Report on Bioavailability of Dioxins and Dioxin-Like Compounds in Soil*. In their report, USEPA identified six studies that reported a total of 17 RBA test results for 2,3,7,8-TCDD in soil and sediment at concentrations ranging from 1.9 to 2,300 pg/kg. These studies reported bioavailability ranging from less than 0.01 to 0.49 (i.e., <1–49 percent). The arithmetic average of the mean bioavailability from each study was 0.23 (i.e., 23 percent). This value represents the "absorbed fraction from exposure medium on site" in Equation 5-5, above, and was divided by the assumed absorbed fraction of 0.50 (i.e., 50 percent) used in establishing toxicity criteria for DLCs adopted for this BHHRA (JECFA 2002). The resulting $RBA_{soil\text{-}sediment}$ was 0.50, and this value was applied to calculation of exposures to all dioxin and furan congeners via incidental ingestion of soil and sediment. Given differences in the behavior of different DLCs in the environment, there is some uncertainty associated with the application of a value based on TCDD to all DLCs.

An $RBA_{soil\text{-}sediment}$ of 0.50 was also adopted for assessment of exposures to arsenic via direct incidental ingestion of soil and sediment. This value was based on the findings of two meta-analyses (USEPA 2010f; Roberts et al. 2007) that reported ranges of bioavailability in soil from 0.05 to 0.31 and from 0.10 to 0.61, respectively. These meta-analyses are summarized below:

- USEPA (2010f) completed in vivo tests of 29 test materials from contaminated arsenic and clean sites using the Juvenile Swine Model. The test materials represented a variety of arsenic phases (e.g., oxides, sulfates, phosphates). Discounting three tests that were determined to be unreliable due to levels of administered arsenic, estimated bioavailability values ranged from less than 0.10 to 0.61 (i.e., 10 to 61 percent) with a mean of 0.34 (i.e., 34 percent). Based on these findings USEPA Region 8 concluded that a RBA of 0.50 as a generally conservative default value for inorganic arsenic (USEPA 2012a).
- Bioavailability studies conducted by Roberts et al. (2007) in cynomolgus monkeys measured the bioavailability of arsenic in 14 soil samples from 12 different sites, including mining and smelting sites, pesticide facilities, cattle dip vat soil, and chemical plant soil. The reported bioavailability ranged from 0.05 to 0.31 (i.e., 5 to 31 percent).

Based on the above studies, the term "absorbed fraction from exposure medium on site" in Equation 5-5 was conservatively assumed to be 0.50. The absorbed fraction from drinking water, which is the dosing medium in the study that provides the basis for the toxicity criteria for inorganic arsenic used for this BHHRA, was assumed to be 1. Therefore the $RBA_{soil\text{-}sediment}$ for arsenic was set to 0.50 for the BHHRA.

A $RBA_{soil\text{-}sediment}$ for all other $COPC_{HS}$ was conservatively assumed to be 1.0. Additionally, the relative bioavailability from tissue ingestion (RBA_{tissue}) was assumed as 1.0 for all $COPC_{HS}$.

Dermal Absorption Factor for Soil and Sediment

The dermal absorption factor represents the proportion of a chemical that is absorbed across the skin from the soil and/or sediment matrix once it has been contacted. Skin permeability is related to the solubility or strength of binding of the chemical in the soil or sediment matrix compared to the skin's stratum corneum and the degree to which the chemical can penetrate the stratum corneum to enter the bloodstream. Therefore, dermal absorption is dependent on the properties of the chemical itself, as well as on external factors including the physical properties of the soil or sediment matrix (e.g., particle size, organic carbon content) and the conditions of the skin (e.g., skin condition, moisture content). Data with which to characterize dermal absorption of chemicals from sediment is not readily available and dermal absorption of chemicals from soil and sediment matrices will differ to some degree. In the absence of sediment-specific information, however, USEPA (2004) supports the application of factors derived for soil to sediment.

Dermal absorption factors for dioxins and furans, arsenic, PCBs, and bis(2-ethylhexyl)-phthalate (BEHP) were obtained from USEPA (2004). Those for chromium, mercury, and nickel were obtained from the California Environmental Protection Agency, Office of Environmental Health Hazard Assessment's (OEHHA) *Technical Support Document for Exposure Assessment and Stochastic Analysis, Draft* (CalEPA 2011). Following USEPA (2004) guidance, in the absence of available data for copper and zinc, a conservative dermal absorption factor of 1.0 was assumed for these $COPC_{HS}$. The dermal absorption factors applied in this BHHRA are presented in Table 5-7.

Chemical Reduction Due to Preparation and Cooking

It is well recognized that preparation and cooking may reduce chemical concentrations in fish tissues, particularly for lipophilic compounds such as dioxins, furans, and PCBs (USEPA 2000b, 2002b; Wilson et al. 1998). These changes are dependent on a number of factors including the lipophilicity of the compound, the type of fish, and the parts of the fish consumed.

For the deterministic CTE and RME evaluations, a cooking loss of 0 (zero percent loss) was conservatively assumed for PCBs and dioxins. In line with the EAM (Appendix A), the impact of applying a cooking loss of 0.25 (25 percent loss) was explored in the uncertainty evaluation for the risk characterization and available information on distributions of cooking loss were considered in the PRA. Following the submittal of the EAM in May 2012, a meta-analysis was published that provided a critical review of the available data on cooking loss factors for lipophilic compounds (AECOM 2012). The findings of this study are also discussed in the uncertainty evaluation.

5.1.2.3 Probabilistic Exposure Evaluation

A probabilistic exposure evaluation was completed for scenarios that met one or more of the following thresholds (Figure 1-4):

(1) The cumulative estimated exposure from all pathways resulted in an incremental cancer risk $>1\times10^{-4}$
(2) The cumulative estimated exposure from all pathways resulted in a total endpoint-specific noncancer HI>1
(3) The cumulative estimated exposure from all pathways resulted in a dioxin cancer HI>1.

The PRA focused on chemicals that were identified as potential risk drivers. Risk drivers were defined as $COPC_{HS}$ that contributed at least five percent of overall risk or hazard across all exposure pathways that made up the selected scenario, and contributed more than 5 percent to the pathway-specific risk or hazard associated with the medium of interest. Both potential exposures within USEPA's Preliminary Site Perimeter and background exposures were evaluated.

Based on the thresholds described above, a PRA was completed for a hypothetical young child fisher and a hypothetical young child recreational visitor. A single model was used to evaluate all hypothetical fishers (i.e., recreational and subsistence). The selection of these receptor groups, as well as the specific scenarios evaluated, are described further in Section 5.2.3.3 of the risk characterization. The general methods, EPCs, and exposure parameters used in the PRA are presented below, with supporting materials provided in Appendix G.

5.1.2.4 General Methods

Probabilistic analyses were completed using Oracle® Crystal Ball software (Gentry et al. 2005). Crystal Ball employs Monte Carlo analysis, a commonly used probabilistic numerical technique where the uncertainty and variability in exposure and resulting hazard/risk estimates are characterized by developing distributions that present the full range of potential exposures.

For the PRA probability distributions were assigned to select exposure parameters to yield an output probability distribution for the exposure estimate rather than a single estimate. A probability distribution is a mathematical function that describes the values and the associated probabilities for a given parameter. For example, there are a wide range of body weights within the human population for a given age group, and the probability distribution for body weight is described as lognormal, which means that it is best represented as a bell-shaped curve with a long tail to the right. The shape of the curve represents the fraction of the population characterized by each body weight, with most individuals clustered together around a fairly limited range of body weights, but with a small number of individuals with a wide range of higher body weights represented by the long tail.

For this evaluation, a 1-dimensional probabilistic analysis, which focused on variability in exposure but did not quantify uncertainties, was completed. The distinction between variability and uncertainty is an important one. Exposure factors vary within the population (e.g., a wide range of fish ingestion rates, exposure durations, body weights), and they can also be uncertain because of a lack of or limited information available about a specific parameter. Parameter variability is an inherent reflection of the natural variation within a population. Uncertainty represents limited or lack of perfect knowledge about specific variables, models, or other factors. Uncertainty can be reduced through further study, measurements, etc., whereas variability cannot. Although the explicit focus of this PRA was to model variability in exposure (and resulting potential risk), all of the distributions used for the PRA inherently also include varying amounts of uncertainty that exist in the exposure parameters.

To develop the output distribution for exposure, the exposure estimate for a receptor–$COPC_H$ pair was repeatedly calculated by Crystal Ball. Each iteration of the exposure model used different combinations of parameter values, as determined by random sampling of the probability distributions for those input parameters that were treated probabilistically (USEPA 2001). For each scenario evaluated, 10,000 simulations were run. A quantitative sensitivity analysis was also performed to test the effect of certain input parameter distributions on the exposure outcome.

5.1.2.4.1 Exposure Point Concentrations

For the PRA, EPCs were established for $COPC_{HS}$ that were identified as potential risk drivers in the scenarios selected for analysis. Specifically, these were dioxins and furans in sediments, soils, and edible tissues; PCBs in all edible tissues; and mercury in catfish fillet. The EPCs were developed as distributions based on the best-fit distribution of the data. For datasets with sample sizes less than 15, the upper-bound for the EPCs for the PRA was established as the mean value plus three standard deviations. For datasets with sample sizes equal to or greater than 15, the maximum concentration in the distribution was established as the maximum detected concentration. This sample size-dependent approach was used because the larger datasets allowed for more complete characterization of the conditions being studied. The lower bound for all distributions was set as a minimum concentration of zero.

5.1.2.4.2 Exposure Parameters

A brief description of the exposure parameters selected for the PRA is provided below. A complete discussion and supporting rationale for each parameter is included in Appendix G. Tables 5-8 and 5-9 provide a summary of the exposure parameters adopted for the PRA and show how they differ from those selected for the deterministic evaluation.

Variable Correlation

Correlation is a measure of the association between two quantitative variables (USEPA 2001). Correlations between variables, whether expressed as single exposure parameter values or a statistical distribution, may be important in a probabilistic model. For example, body weight and ingestion rate may be correlated (e.g., children with a higher body weight may ingest more fish compared to children with a lower body weight). The dependence and a quantitative relationship between other parameters, including body weight and surface area, are well established.

With the exception of body weight and surface area (discussed further below), all exposure parameters were assumed to act independently: that is, a mechanism to account for the correlations between other parameters was not incorporated into the PRA model. This is because no specific evidence or data were found which could serve as the basis for a quantitative correlation for any other pairs of parameters used in the PRA models. The impact of this decision is discussed within the context of the results for the PRA.

Common Parameters

For the hypothetical young child receptor's exposure duration, a triangular distribution[24] with a minimum of 1 year, most likely value of 3.5 years, and maximum of 6 years was assumed. This distribution was based on best professional judgment with the maximum value set to the RME exposure duration used for the hypothetical young child in the deterministic evaluation. The averaging time for each iteration of the model was set equal to the randomly selected exposure duration for that iteration.

For body weight, a lognormal distribution with a mean of 17.27 kg and a standard deviation of 4.97 kg was used. This relationship was derived by Portier et al. (2007) for children ages 1 through 6 years, and is based on NHANES IV data. The distribution for body weight was bound at the lower and upper ends based on best professional judgment and using lower and upper percentiles of body weight for the defined population.

Parameters for Tissue Ingestion

The assumed input distributions for the fish and shellfish consumption rates for young children were the empirical data collected during the Lavaca Bay (Alcoa 1998) survey upon which the fish and shellfish ingestion rates for the deterministic evaluation were based. It is noted that in this study a large percentage of children who consumed fish during the survey period did not consume any shellfish. Because these individuals were fish consumers, the report on Lavaca Bay included zero values for shellfish ingestion rates for these individuals when calculating ingestion rates for shellfish consumers. The same approach was used for the PRA.

[24] A triangular distribution is a continuous probability distribution with a lower limit, an upper limit and a single modal (i.e., most likely) value. The selection of a value between the straight lines that connect the minimum and modal values and the maximum and modal values is defined by the probability between these two values. These distributions are used when one has information about the range of potential values and a reasonable estimate of the most likely value for that parameter.

The fractions of total fish and shellfish consumed that were harvested from the within USEPA's Preliminary Site Perimeter are likely to vary substantially among individuals. For the PRA, these parameters were both set to a triangular distribution with a most likely value of 0.25, a minimum value of 0.01, and a maximum value of 1. The reported range was based on the findings from the Lavaca Bay study, which were also used in developing the deterministic parameter values for this term, as well as best professional judgment.

Parameters for Direct Contact

The fraction of total intake that was soil versus sediment for each scenario was set to the point estimate that was adopted for the deterministic evaluation and so was not treated probabilistically. The fisher was assumed to be exposed 100 percent of the time to sediment, with no exposure to soils, whereas the recreational visitor was assumed to receive 50 percent of total daily exposure through soil and 50 percent through sediment.

For soil and sediment ingestion rates, a lognormal distribution with a geometric mean of 24 mg/day and geometric standard deviation of 4 mg/day was used. This distribution was based on a long-term estimate of soil ingestion developed from a tracer-element study of 64 children from Anaconda, Montana, and was consistent with other distributions established in the literature (Stanek et al. 2001). A high-end ingestion rate of 1,000 mg/day recommended by USEPA (2011a) for pica behavior was applied as the maximum rate. A minimum ingestion rate of 0 mg/day was used to avoid the possibility of negative ingestion rates.

The exposed surface area for the hypothetical young child receptors was calculated as the product of the total surface area of an individual and the percent of surface area exposed, as follows:

$$SA_{exposed} = SA_{total} \times \%\ \text{surface area exposed} \qquad \text{(Eq. 5-6)}$$

Where:

$SA_{exposed}$ = exposed surface area

SA_{total} = total surface area

The total surface area was calculated as a function of the body weight using the relationship established by Burmaster (1997). The factor for percent surface area exposed was modeled as a range, representing various combinations of the arms, legs, and feet exposed. The factor was assigned a triangular distribution with the most likely value equal to the percentage of total surface area for face, forearms, hands, lower legs, and feet. The minimum value assumed that only the face, forearms, and hands were exposed, while the maximum value assumed that the face, entire arm, hands, entire leg, and feet were exposed. Surface area data were obtained from USEPA (2011a).

For the sediment adherence factor, a uniform distribution[25] with a minimum of 0.09 mg/cm^2 and maximum of 3.6 mg/cm^2 was used. The maximum value assumed was based on body part-specific adherence factors from a study of children playing in tidal flats, weighted to the most likely exposed body parts discussed above. This value was also used in the deterministic evaluation for this BHHRA. In the absence of specific data on adherence to sediments with characteristics similar to those within USEPA's Preliminary Site Perimeter (i.e., fine grained), a minimum value was selected from a study that measured soil adherence in children. In this instance, the range of values represented both variability and uncertainty in the adherence of sediment that could occur. A distribution for the soil adherence factor was not developed. For the PRA, this parameter was treated as a point estimate of 0.09 mg/cm^2 and was the same value that was selected for the deterministic evaluation.

Two distributions of potential exposure frequencies for direct contact with soils and sediments were established—one for the fisher and one for the recreational visitor. The selected values were centered around the factors adopted for the deterministic risk calculation and were developed using best professional judgment. For the potential young fisher, a triangular distribution with a most likely value of 13 days/year, a minimum of 1 day/year, and a maximum of 156 days/year was adopted. For the potential recreational visitor, a triangular distribution with a most likely value of 52 days/year, a minimum of 1 day/year, and a maximum value of 156 days/year was adopted.

[25] A uniform distribution is a straight line, defined by a minimum and maximum value, with an equal probability of selecting any value between the minimum and maximum values. It is used when a reasonable estimate of the range of likely values can be made, but has little information on the probabilities of values between the minimum and maximum.

The distribution for fractional intake of soils and sediments that is related to potential exposures within the area of study was centered around the values assumed for the deterministic evaluation. For the hypothetical young child recreational visitor, a triangular distribution with a most likely value of 0.5, and minimum and maximum values of 0.1 and 1.0, respectively, was adopted. It is possible that a fisher would spend a greater duration of time in locations within the area of study on any given day compared to a recreational visitor. Therefore, for the PRA, a higher fractional intake was adopted for the fisher than for the visitor. For this receptor, a triangular distribution with a most likely and maximum value of 1.0 and a minimum value of 0.5 was assumed.

Chemical Specific Factors

Potential risk-driving COPCHS identified in the deterministic risk assessment were carried forward for further evaluation in the PRA. These were determined to be dioxins and furans in sediment, all tissue types, and in soil; PCBs in all tissue types; and mercury in catfish only.

For the PRA, distributions for chemical reduction due to preparation and cooking were developed for dioxins, furans, and total PCBs. These distributions were based on a meta-analysis of cooking loss studies and findings completed by AECOM (2012). This meta-analysis identified studies, completed in a variety of tissue types, and applied a range of preparation and cooking methods, with sufficient data for quantitative analysis to determine the range and midpoint of cooking loss for dioxins and PCBs. The analysis focused on studies that used a relevant and appropriate experimental method and presented changes in raw and cooked fish tissue COPC levels on a mass basis. This is because a comparison of concentrations in raw and cooked fish alone neglects the change in tissue mass that occurs with cooking, which is often significant. The authors reported percentiles and statistics for cooking loss for dioxins and furans and PCBs. These were used to develop distributions for the cooking loss term for the PRA. The complete distributions are described in detail in Appendix G.

The loss parameters were applied to catfish fillet only and not to clams or crabs. No data on chemical reduction due to preparation and cooking specific to shellfish could be located. Clam tissue analyzed from within USEPA's Preliminary Site Perimeter had a substantially lower percent lipid than most finfish and techniques used for preparing and cooking shellfish differ from those used for finfish. As a result, the application of a loss factor based on cooking loss in finfish was not considered appropriate for shellfish. Therefore for the PRA, the cooking loss for shellfish ingestion was conservatively estimated at 0 percent.

For the oral $RBA_{soil\text{-}sediment}$ for dioxins and furans, a lognormal distribution with a geometric mean value of 0.6 and standard deviation of 0.28, with minimum and maximum values of 0 and 1.0, respectively, was assumed. This distribution was developed using the studies presented by USEPA's (2010d) *Final Report on Bioavailability of Dioxins and Dioxin-Like Compounds*. For the dermal absorption factor for soil/sediment ($ABS_{d\ soil\text{-}sediment}$) for dioxins and furans a uniform distribution with a minimum value of 0.01 and a maximum value of 0.03 was adopted. This distribution was based on USEPA (2004) and studies published by Roy et al. (2008) and Shu et al. (1988).

5.2 Risk Characterization

Risk characterization is the final step in the risk assessment process. In this step, information from previous steps of the risk assessment is synthesized to provide an overall assessment of potential risks associated with the area being studied. The goal of risk characterization is to present and interpret the key findings of the risk assessment, along with their limitations and uncertainties, for use in risk management decision-making.

Cancer and noncancer hazards and cancer risks were quantified by combining the intakes estimated in the exposure assessment with the toxicological criteria compiled in the toxicity assessment to yield numerical estimates of potential health risk for specific receptor types under hypothetical exposure scenarios. A general description of the methods used for combining estimates of exposure and toxicological criteria and interpreting the resulting metrics is presented below. This is followed by the results for the risk characterization for this BHHRA for the area north of I-10 and aquatic environment.

5.2.1 General Methods for Risk Characterization

Three categories of potential health effects were evaluated for this BHHRA: cancer risk, noncancer hazard, and dioxin cancer hazard. The general methods for calculating each is described below.

5.2.1.1 Cancer Risk

For all carcinogenic $COPC_{HS}$ other than dioxins and furans, cancer risk estimates were derived using standard risk assessment methods that estimate the incremental probability that an individual described by hypothetical exposure scenarios might develop cancer during his or her lifetime as a result of exposure to COPCs in the area under study. The term "incremental" reflects the fact that the calculated risk associated with any exposures in the area under study is in addition to the background risk of cancer experienced by all individuals in the course of daily life; that is, any risks associated with any exposures in the area under study are considered to be an incremental increase in the probability of developing cancer in addition to the background probability that an individual might develop cancer during his or her lifetime.

Excess incremental lifetime cancer risks were calculated as the product of the estimated exposure (i.e., LADD) and the expression of the carcinogenic potency of chemicals (e.g., cancer slope factor [CSF]). Excess incremental lifetime cancer risk from oral and dermal exposures was calculated using the following equation:

$$Cancer\ \ Risk(unitless) = LADD \times CSF \qquad \text{(Eq. 5-7)}$$

Where:

LADD	=	lifetime average daily dose of the chemical via the specified exposure route (mg/kg-day)
CSF	=	cancer slope factor (kg-day/mg).

For each hypothetical receptor and exposure scenario, incremental cancer risks were summed across all the exposure pathways for each chemical and then across chemicals to estimate overall incremental cancer risk.

Both federal and state regulatory agencies define what they consider to be an acceptable level of incremental cancer risk associated with exposure to chemicals in environmental media. USEPA considers 1×10^{-6} to 1×10^{-4} the target range for excess cancer risk (USEPA 1990).

The potential for cancer from exposure to dioxins and furans was evaluated as "dioxin cancer hazard." This process is described in Section 5.2.1.3 below.

5.2.1.2 Noncancer Hazard

Noncancer health risks are termed hazards. When an HI exceeds 1, this indicates that under the hypothetical exposure scenario evaluated, there is some potential for adverse health effects to occur as a result of chemical exposures assumed to have occurred in the area under study based on the hypothetical receptor and exposure scenario. To evaluate noncancer hazards, the ratio of the exposure term (i.e., average daily dose) to the corresponding noncancer toxicity reference value (i.e., RfD) is calculated. The HQ is calculated for each exposure route using the following equation:

$$HQ(unitless) = \frac{ADD}{RfD} \qquad \text{(Eq. 5-8)}$$

Where:

ADD = average daily dose of the chemical via the specified exposure route (mg/kg-day)

RfD = reference dose (mg/kg-day).

To evaluate the effect of exposure via multiple exposure routes for each receptor, the route-specific HQs are summed for each $COPC_H$ to determine a noncancer HI using the following formula:

$$HI(unitless) = HQ_1 + HQ_2 + ... + HQ_i \qquad \text{(Eq. 5-9)}$$

Where:

HI = hazard index

HQ = hazard quotient for a specified exposure route (unitless).

Once the HQs for individual $COPC_H$s were summed for an individual receptor to derive a $COPC_H$-specific HI, the $COPC_H$-specific HIs were summed to derive a total HI for that exposure scenario.

HIs that are calculated for multiple chemicals as described above are likely to overstate risk if the RfDs for the chemicals are based on adverse effects on different target organs. This is because the noncancer health hazards associated with chemicals that affect different target organs or have different health effects are not likely to be additive. For this BHHRA, following USEPA guidance (1989) in the case that the total HI for a receptor exceeded 1 for all $COPC_H$s combined, separate hazard indices for group of $COPC_H$s that affect the same target organ or endpoint were estimated. These effect-specific HIs provide a more accurate indication of whether there is potential for a specific adverse health effect to occur for a specific hypothetical receptor and exposure scenario.

If the resulting multi-chemical or effect-specific HI is less than 1 for a given hypothetical exposure scenario, then no adverse health effects are expected to occur (USEPA 1989). If the HI is greater than 1, then further risk evaluation may be appropriate. However, HIs greater than 1 do not necessarily mean that any actual adverse health effects would be observed in a receptor population under the hypothetical exposure scenario that provides the basis for the exposure estimate. A substantial margin of safety has been incorporated into the RfDs developed for the COPCs. For these chemicals, adverse health effects may not occur even if the HI is much larger than 1. The ratio is not a measure of probability that adverse health effects will occur. That is, the level of concern for health effects to occur does not necessarily increase linearly as the RfD is approached or exceeded (USEPA 1989).

5.2.1.3 Dioxin Cancer Hazard

As discussed in the TESM (Appendix B), the scientific literature indicates that dioxins act via a non-linear mode of action, which suggests that a threshold dose must be reached before a carcinogenic effect can occur (Integral 2012b). Consistent with this concept, the carcinogenic potential for TEQ_{DF} was estimated for this BHHRA using a hazard metric like that described for noncancer hazards above. Cancer hazards due to TEQ_{DF} were expressed as an HQ for a single potential exposure route and an HI when hazards from all potential exposure routes for a receptor were summed. Because cancer is a different toxic endpoint from the noncancer endpoints, the HIs for dioxin were not summed with noncancer hazards.

5.2.1.4 Age Groups and Exposure Durations

Cancer risks, noncancer hazards, and dioxin cancer hazards were characterized for different age groups. As is customary in the practice of human health risk assessment, cancer risks for nonthreshold carcinogens were evaluated over a lifetime using the LADD as the intake metric. For this estimate, the intake for each individual age group was calculated and intakes for all relevant age groups were combined and summed to derive a total LADD; for the RME hypothetical recreational fisher, subsistence fisher, and recreational visitor, six years of exposure as a young child, 11 years of exposure as an older child, and 16 years of exposure as an adult were assumed and summed to estimate exposure for a total combined exposure period of 33 years. For the CTE hypothetical recreational fisher, subsistence fisher, and recreational fisher, only exposure to an adult was evaluated for the total assumed exposure duration of 33 years.

In contrast to cancer risks, noncancer and cancer hazards for threshold carcinogens are generally estimated separately for life stages for which differences in behavior and relative intake (per unit body weight) are exhibited. Because intake for noncancer hazards is estimated using an average daily dose rather than the LADD used to evaluate cancer risk, the life stage that results in the highest potential exposure for an individual will also exhibit the greatest potential hazard. For this BHHRA, noncancer hazards and dioxin cancer hazards were estimated for the age group that had the highest relative potential exposure of all age groups conceptualized for a given scenario. For all RME scenarios, this was the young child. A comparison of the potential pathway-specific RME doses for each age group is presented in

Table 5-10. These ratios were calculated using the exposure parameters presented in Table 5-6. CTE hazards were estimated for an adult.

5.2.2 Deterministic Risk Assessment

This section presents the baseline deterministic risk results by potential receptor group. A summary of all estimated RME hazards and risks is provided in Table 5-11; a summary of estimated CTE hazards and risks is provided in Table 5-12. The full set of risk and hazard estimates are provided as Appendix H. Tables H-1 through H-14 present assumed exposures, and resulting estimated hazards and risks by exposure medium. Tables H-15 through H-42 present estimated hazards and risk by exposure scenario. Tables H-43 through H-54 show the contribution of each $COPC_H$ and each exposure pathway to overall risks and/or hazards for the hypothetical scenarios that resulted in excess cancer risk above 1×10^{-4} or hazards greater than 1. These relative contributions were used for identifying risk drivers.

5.2.2.1 Hypothetical Recreational Fisher

Potential exposure routes for hypothetical recreational fishers were assumed to include incidental ingestion and dermal contact with sediment and ingestion of fish or shellfish. Twelve hypothetical exposure scenarios were evaluated for this receptor group. These included direct contact exposure at one of four beach areas (A, B/C, D, or E) in combination with the ingestion of catfish fillet, crabs, or clams from the adjacent FCA evaluated for the particular type of tissue.

Table 5-13 presents a summary of estimated cumulative noncancer hazard, cancer risk, and dioxin cancer hazard for the hypothetical recreational fisher scenarios. The noncancer RME HIs ranged from 0.03 to 50, while the CTE HIs were all less than 1. Table 5-14 presents endpoint-specific HIs for all hypothetical recreational fishing scenarios that exhibited a HI greater than 1. Three scenarios with an overall HI greater than 1 did not exhibit any endpoint-specific HI greater than 1, including: 1) Scenario 1A – Direct contact with sediment at Beach Area A and ingestion of catfish from FCA 2/3; 2) Scenario 2A – Direct contact with sediment at Beach Area B/C and ingestion of catfish from FCA 2/3; and 3) Scenario 4A – Direct contact with sediment at Beach Area D and ingestion of catfish from

FCA 1. The noncancer hazards associated with these hypothetical scenarios are not discussed further.

The only hypothetical recreational fisher scenarios that had endpoint-specific RME HIs greater than 1 were those that assumed potential direct contact with sediments at Beach Area E. For these scenarios, the vast majority of the estimated noncancer hazard was attributable to direct exposure to sediment (Appendix H). For hypothetical recreational fisher scenarios assuming exposure at Beach Area E and consumption of either crabs or clams, only the HI specific to reproductive/developmental endpoints exceeded 1, and TEQ_{DF} intake contributed 99 percent of the estimated hazard. For hypothetical recreational fisher scenarios assuming exposures at Beach Area E and the consumption of catfish from FCA 2/3 (i.e., Scenario 3A), HIs specific to reproductive/developmental endpoints and immunotoxity endpoints both exceeded 1 and were estimated at 40 and 2, respectively. For exposures assumed to occur under the conditions defined by this scenario, TEQ_{DF} contributed 98 percent of the reproductive/developmental HI, and mercury contributed the remaining 2 percent. The HI specific to immunotoxicity was primarily influenced by PCBs in catfish (Appendix H).

Across all hypothetical recreational fisher scenarios, cumulative estimated RME cancer risks ranged from 5×10^{-7} to 2×10^{-5}. Cumulative estimated CTE cancer risks were more than an order of magnitude lower and ranged from 2×10^{-8} to 7×10^{-7} (Table 5-13).

TEQ_{DF} cancer HIs were all less than 1 for hypothetical recreational fisher exposure scenarios assuming direct contact at Beach Area A, B/C, or D, and the consumption of catfish fillet, crabs, or clams from the adjacent FCA. For hypothetical recreational fisher scenarios that assume direct contact at Beach Area E and ingestion of catfish from FCA 2/3, crabs from FCA 2, or clams from FCA 2/3, the RME cancer HIs for TEQ_{DF} were all 10. For these scenarios, assumed, direct contact with sediments contributed over 98 percent of the total hazard (Appendix H). The estimated CTE TEQ_{DF} cancer HI was less than 1 for all hypothetical recreational fisher scenarios.

Overall, hypothetical recreational fisher scenarios that assumed direct contact at Beach Area E were the only scenarios that resulted in endpoint-specific noncancer HIs and TEQ_{DF}

cancer HIs greater than 1. No cumulative cancer risks for these scenarios exceeded the 1×10^{-4} threshold (Table 5-13). Direct contact assumed to occur at Beach Area E accounted for over 98 percent of the hazards for hypothetical recreational fisher scenarios. Assumed exposure to TEQ_{DF} contributed 98 percent of the estimated hazard for these direct pathways. For catfish consumers, PCBs in catfish, in combination with assumed direct exposure at Beach Area E, contributed to hazards in Scenario 3A (Table 5-14).

It is important to note when considering the risk results, Beach Area E was capped as part of the TCRA, and that any potential direct contact exposure to sediments in this area are no longer possible under current, post-TCRA conditions. The implication of limiting exposure to the sediments present within the 1966 perimeter of the northern impoundments was evaluated in Appendix F, and discussed further in Section 5.2.3.2.

5.2.2.2 Hypothetical Subsistence Fisher

The exposure pathways and scenarios that were evaluated for hypothetical subsistence fishers were identical to those evaluated for the hypothetical recreational fisher. The differences between the hypothetical subsistence and recreational fisher scenarios were the frequency and intensity with which each receptor group was assumed to be exposed to sediments in the area under study and the amount of finfish or shellfish tissue that each was assumed to consume. This second factor was a result of variations in the parameters incorporated for both the total ingestion rate assumed for finfish and shellfish and the fractional intake of finfish and shellfish that was assumed to come from the area under study. Because subsistence fishing is defined as a high-end exposure, no CTE risks or hazards were estimated for this potential receptor group.

As for the hypothetical recreational fisher, twelve separate exposure scenarios were assumed and evaluated for the hypothetical subsistence fisher. These included direct contact at one of each of the four beach areas in combination with the ingestion of catfish fillet, crabs, or clams from the appropriate, adjacent FCA.

Table 5-15 presents a summary of estimated cumulative noncancer hazards, cancer risks, and TEQ_{DF} cancer hazards for the hypothetical subsistence fisher scenarios. Although overall hazards and risks were greater for the hypothetical subsistence fisher scenario than for the hypothetical recreational fisher scenario, similar trends in the relative risks associated with the various exposure units and the contribution of specific $COPC_{HS}$ to overall hazards and risks were observed.

Across all hypothetical subsistence fisher scenarios evaluated, the overall noncancer RME HI ranged from 0.2 to 100 (Table 5-15). The noncancer HIs for these scenarios were 2 to 11 times greater than the RME HIs for the hypothetical recreational fisher scenarios, and more than an order of magnitude greater than the CTE HIs estimated for adults under the hypothetical recreational fisher scenario.

Table 5-16 presents endpoint-specific noncancer HIs for all hypothetical subsistence fisher scenarios with an overall HI greater than 1. As was the case for the scenarios evaluated for the recreational fisher, the greatest noncancer hazards were estimated for the hypothetical subsistence fisher scenarios that assumed direct contact with sediments at Beach Area E under the baseline condition (i.e., immediately prior to the TCRA). The reproductive/developmental-specific HI associated with the assumed direct contact at Beach Area E (Scenario 3A, Table 5-16) was 100 for subsistence fishers. Assumed direct contact with sediments alone at other beach areas did not result in overall noncancer HIs greater than 1 (Appendix H). Unlike the hypothetical recreational fisher, assumed consumption of fish and shellfish from certain FCAs in the subsistence fisher scenario, resulted in endpoint-specific noncancer HIs that were greater than 1 (e.g., Scenario 2A), even without direct contact with beach sediments. Noncancer hazards from the assumed ingestion of catfish from either FCA 2/3 or FCA 1 were largely influenced by TEQ_{DF} (47 percent of overall hazard), PCBs (38 percent of overall hazard), and mercury (12 percent of overall hazard). Hazards from the assumed ingestion of clams from FCA 2 were largely influenced by TEQ_{DF} (90 percent of overall hazard) and PCBs (9 percent of overall hazard) (Appendix H).

Across all hypothetical subsistence fisher scenarios, cumulative RME excess cancer risks ranged from 3×10^{-6} to 1×10^{-4} (Table 5-15) and, thus, fell within EPA's target risk range.

The TEQ_{DF} cancer HI for hypothetical subsistence fisher Scenarios 3A, 3B, and 3C, all of which assumed direct contact with sediments in Beach Area E under the baseline condition, was 40 (Table 5-15). In addition, the hypothetical subsistence fisher scenarios that assumed direct contact with other beach areas and the consumption of catfish from FCA 2/3 or FCA 1 resulted in a TEQ_{DF} cancer HI of 3. The TEQ_{DF} cancer hazards for hypothetical subsistence fisher scenarios that assumed direct contact at Beach Area A, B/C, or D, and which assumed the ingestion of crabs or clams from the adjacent FCAs, were all less than 1.

5.2.2.3 Hypothetical Recreational Visitor

Exposure routes assumed for the hypothetical recreational visitor scenario included assumed incidental ingestion and dermal contact with a combination of soil and sediment. Four hypothetical exposure scenarios were evaluated for this receptor; these assumed direct contact with sediments at each of the four beach areas combined with direct contact with soils throughout the northern impoundments.

Table 5-17 presents a summary of cumulative noncancer hazards, cancer risks, and TEQ_{DF} cancer hazards for the recreational visitor scenarios. Details on noncancer hazards are presented in Table 5-18.

The hypothetical recreational visitor scenario, which assumed baseline exposure via direct contact with sediments at Beach Area E and soils throughout the area north of I-10 (Scenario 3), resulted in the highest noncancer hazards, excess cancer risks, and TEQ_{DF} cancer hazard. For this scenario, the overall RME noncancer HI was 60, and over 99 percent of that hazard was attributable to exposure to TEQ_{DF} in sediments at Beach Area E (Appendix H). The CTE noncancer hazard was less than 1. For hypothetical recreational visitor scenarios assuming direct contact with sediments at Beach Area A, B/C, or D and soils throughout the northern impoundments, the resulting noncancer RME HIs were all less than 1.

Table 5-18 presents endpoint-specific HIs for hypothetical recreational visitor scenarios. The only hypothetical recreational visitor scenario that resulted in an RME noncancer HI greater than 1 was the scenario that assumed direct contact with sediments in Beach Area E and the soils in the impoundments north of I-10. The hazards associated with this scenario were largely attributed to reproductive/developmental endpoints, and the HI for this specific endpoint was equal to the overall HI at 60. No other endpoint-specific HIs were greater than 1 for this scenario.

For all hypothetical recreational visitor scenarios, cumulative RME cancer risks ranged from 8×10^{-7} to 1×10^{-5} (Table 5-17). Cumulative CTE cancer risks were more than an order of magnitude lower.

The hypothetical recreational visitor scenario that assumed direct contact with sediments at Beach Area E and soils north of I-10 was estimated to have an RME TEQ_{DF} cancer HI of 20 (Table 5-17). The corresponding CTE TEQ_{DF} cancer HI for this scenario was less than 1. As for the noncancer effects, over 99 percent of the cancer hazard was attributable to assumed exposure to sediments at Beach Area E. For hypothetical recreational visitors exposed to other beach areas, in combination with soils north of I-10, the RME and CTE cancer TEQ_{DF} HIs were all less than 1.

5.2.3 Refined Analyses

Consistent with the approach summarized in Figure 1-4, additional analyses were completed to further characterize risks and/or hazards estimated for the hypothetical exposure scenarios that met one or more of the following thresholds:

(1) The cumulative estimated exposure from all pathways resulted in excess cancer risk $>1\times10^{-4}$
(2) The cumulative estimated exposure from all pathways resulted in a total endpoint-specific noncancer HI>1
(3) The cumulative estimated exposure from all pathways resulted in a dioxin cancer HI>1.

Although none of the scenarios included in the baseline deterministic evaluation resulted in an estimated cancer risk greater than 1×10^{-4}, certain hypothetical scenarios resulted in endpoint specific HIs greater than 1 or dioxin cancer HIs greater than 1. Table 5-19 presents a summary of these scenarios. The refined analyses for each selected scenario consisted of three evaluations: 1) an analysis and comparison of background hazards with the estimated deterministic hazards for the area under study, 2) an evaluation of post-TCRA hazards, and 3) a PRA of potential hazards.

5.2.3.1 Background Hazard Evaluation

Background hazards for exposure routes that compose the scenarios selected for refined analysis (Table 5-19) were calculated using the same assumptions about frequency and duration of exposure to each medium as were used in the main analysis of risks for USEPA's Preliminary Site Perimeter. Resulting exposures, hazards, and risks were tabulated (Appendix I). These results were then compared to corresponding results of the deterministic baseline evaluation for the area under study. The background noncancer hazards and dioxin cancer hazards are provided in Appendix I.

Estimated background RME and CTE noncancer hazards and dioxin cancer hazards for hypothetical recreational fishers, subsistence fishers, and recreational visitors are provided in Tables 5-20 and 5-21, respectively. To compare estimated baseline and background exposures, the RME noncancer hazard and dioxin cancer hazard endpoints were emphasized because these were the only endpoints for which the RME HIs in the baseline deterministic evaluation exceeded the target of 1.

Using background concentrations, the hypothetical subsistence fisher scenario that assumed the consumption of catfish was the only scenario that resulted in noncancer HI greater than 1 (Table 5-20). While the risks for the area under study were higher than background risks, it is important to note that background conditions resulted in a noncancer HI of 10 under this scenario. It is also useful to compare the estimated hazards that result from estimated exposure to each individual medium, so that the importance of each medium, and its contribution to risks for the area under study, relative to background and under baseline conditions, can be better understood.

Below, the absolute differences in assumed exposures and resulting hazards for the area under study and background are presented for each individual exposure medium, but are not presented for each receptor type. This is because this relative difference is the same for all scenarios. For example, if exposure to unit 1 under the hypothetical recreational fisher scenario resulted in twice the hazard estimated for this scenario in unit 2, then the same relative difference in exposure and risk was also true for the hypothetical subsistence fisher scenario evaluated in unit 1 versus unit 2.

5.2.3.1.1 Direct Contact with Sediment

The endpoint-specific RME noncancer HIs and the cancer TEQ_{DF} HIs for baseline exposure via direct contact with sediments at Beach Area E were greater than 1 for all hypothetical scenarios evaluated (Table 5-19). For the hypothetical recreational fisher scenario with assumed exposure via direct contact at Beach Area E, the RME noncancer TEQ_{DF} HI was 50 (ingestion and dermal contact combined) and the RME cancer TEQ_{DF} HI was 10. More than 98 percent of the total noncancer hazard was attributable to TEQ_{DF} (Table 5-11).

Hazards associated with dioxins and furans in background shoreline sediments were substantially lower (Table 5-20). Under identical exposure conditions for the hypothetical recreational fisher scenario identified above, but using background sediment concentrations, the RME noncancer TEQ_{DF} HI was only 0.02, and the RME TEQ_{DF} cancer HI was 0.0006 (Table 5-20). Therefore, risks for the hypothetical recreational fisher scenario, when assumed exposures included contact with beach sediment in background areas, were less than 1 percent of those calculated for assumed direct contact with sediments at Beach Area E under this scenario.

The RME noncancer TEQ_{DF} HI for the hypothetical recreational fisher scenario, including exposure to sediments in other beach areas on the Site (i.e., Beach Area A, B/C, or D) ranged from 0.01 (for Beach Area A) to 0.07 (for Beach Area B/C). These were comparable to the estimated background risks. The RME cancer TEQ_{DF} HIs for direct exposure to sediments in these beach areas (excluding Beach Area E), which ranged from 0.0005 to 0.002, were also comparable to background HIs (i.e., range of TEQ_{DF} HIs of 0.0006 to 0.002).

Based on this analysis, it appears that potential risks due to direct contact with sediment in all areas except Beach Area E, were comparable to background risks. Potential risks in Beach Area E, under the baseline condition (i.e., immediately prior to the TCRA), exceeded background risks.

5.2.3.1.2 Catfish Ingestion

Assumed ingestion of catfish from the area under study resulted in RME noncancer HIs that were greater than 1 for all fishing scenarios; the cancer TEQ_{DF} HIs were greater than 1 for only the hypothetical subsistence fisher scenario. Figure 5-6 shows RME noncancer HQs, by $COPC_H$, for assumed consumption of catfish from FCA 2/3, FCA 1, and background for both the hypothetical recreational and subsistence fisher scenarios. TEQ_{DF}, PCBs, and mercury were the largest contributors to total noncancer hazards associated with assumed consumption of catfish at the area under study (Figure 5-6).

For TEQ_{DF} and PCBs, the estimated hazards resulting from ingestion of catfish from FCA 1 and FCA 2/3 were greater than the hazard associated with ingestion of catfish containing background levels of these $COPC_{HS}$. It is important to note, however, that 41 to 42 percent of the baseline hazard attributed to TEQ_{DF}, and 55 to 60 percent of baseline hazard associated with PCBs were also present under background conditions. Hazards associated with exposure to methylmercury in catfish fillets were higher for background than for FCA 2/3 and were comparable to background for FCA 1.

This analysis indicates that while risks associated with the assumed consumption of catfish from the area under study were higher than background risks, background levels of TEQ_{DF} and PCBs contributed substantially to total risk estimates. For mercury, the estimated background risks were similar to or exceeded the risks associated with the area under study, indicating that the area under study is not contributing additional risks for this $COPC_H$.

5.2.3.1.3 Ingestion of Clams

Assumed consumption of clams from FCA 2 resulted in RME endpoint-specific noncancer HIs that exceeded 1 for the hypothetical subsistence fisher scenario. When combined with other exposure pathways, the consumption of clams contributed to cumulative noncancer hazards that were greater than 1 for the hypothetical recreational fisher scenario. (As discussed previously, the vast majority of the noncancer hazard under this scenario was from assumed direct contact with sediment in Beach Area E [Table 5-11]). Assumed consumption of clams from FCA 2 also contributed to dioxin cancer hazards that were greater than 1 when all exposure pathways were summed for both the hypothetical recreational and subsistence fisher scenarios. Figure 5-7 presents noncancer HQs, by $COPC_H$, for consumption of clams, calculated using $COPC_H$ concentrations for exposure units within USEPA's Preliminary Site Perimeter and background.

Although no cumulative hazards for scenarios that assumed consumption of clams from FCA 1/3 resulted in a HI greater than 1, the noncancer HQs that were estimated from concentrations in clams from this FCA were included to provide additional perspective on the impact of background levels. As illustrated in Figure 5-7, the contribution of clam consumption to the cumulative hazard quotient for the hypothetical recreational and subsistence fisher scenarios was much larger for FCA 2 than for either FCA 1/3 or background (Figure 5-7), and tissue concentrations of TEQ_{DF} in clam were the largest driver for these differences.

5.2.3.1.4 Ingestion of Crabs and Direct Contact with Soil

Ingestion of crabs and direct contact with soils were minor contributors to scenarios that resulted in HIs greater than 1. Although the assumed consumption of crabs from FCA 1 contributed to cumulative TEQ_{DF} noncancer and cancer HIs that exceeded 1 for hypothetical Scenario 3C (i.e., direct contact with sediments at Beach Area E and the consumption of crab from FCA 1), consumption of crab itself, did not result in a HI greater than 1 for any scenario evaluated, and it contributed less than 1 percent of the total HI reported for Scenario 3C (Appendix H). Similarly, direct contact with soils in the area north of I-10 contributed less than one percent to the cumulative noncancer and dioxin cancer HIs for Scenario 3 (i.e., Scenario 3—Direct contact with sediments at Beach Area E and soils north of I-10). Given the minor contributions of crab ingestion and direct contact with soil to hazard estimates for the area under study, a discussion of background hazards associated with these exposure pathways is not presented.

5.2.3.1.5 Summary of Comparisons of Baseline Risks to Background

Background concentrations of certain COPCs contributed substantially to potential risks associated with certain media. Hypothetical baseline exposure to sediments in Beach Area E resulted in potential risks that exceeded background levels, but the risks estimated for sediments in the other beach areas within USEPA's Preliminary Site Perimeter (Beach Areas A, B/C, and D), were consistent with risks calculated using background concentrations, indicating that potential risks due to sediments in those areas are not elevated above background levels.

Assumed ingestion of catfish from the area under study resulted in higher potential risks than ingestion of catfish from background locations. However, background concentrations contributed substantially to total risks, providing roughly one-half of the total risks estimated for PCBs and TEQ_{DF}. In addition, the background analysis indicated that all of the risks associated with mercury in catfish were likely due to background concentrations of mercury.

While the assumed consumption of clams did not contribute substantially to total risks, the analysis of background indicated that risks associated with the consumption of clams from FCA 2 exceeded background risks and resulted in a pathway-specific HQ greater than one for the hypothetical subsistence fisher scenario. Risks associated with assumed consumption of clams from FCA 1/3 were slightly higher than background risks, but contributed only marginally to the cumulative hazard for both hypothetical recreational and subsistence fishers in comparison to assumed direct exposure to sediment at Beach Area E.

Direct contact with soils in the area of study and ingestion of crab did not contribute substantially to total estimated risks for those hypothetical scenarios that assumed these routes of exposure and exceeded a HI of 1. Therefore, an analysis of background risks was not conducted for these media.

5.2.3.2 Post-TCRA Evaluation

An evaluation of post-TCRA noncancer hazards and dioxin cancer hazards was completed for the scenarios outlined in Table 5-19. The post-TCRA exposures considered for the sediments and soils reflect the limited access of individuals to large portions of the area within USEPA's Preliminary Site Perimeter, as a result of the implementation of the TCRA (Figure 1-3). As described in Appendix F, the post-TCRA evaluation also incorporates model-estimated reductions in the concentrations of dioxins and furans in catfish tissue and the exclusion of clam tissue from Transect 3 from the dataset used to calculate clam EPCs; for crab, no change in tissue concentrations from baseline conditions was assumed. Both the hazards associated with the post-TCRA condition, as well as a measure of the reduction in hazard resulting from implementation of the TCRA were evaluated. Hazard reduction for the area under study was defined as the percentage of such hazard (i.e., indicated as baseline hazard above background) that was removed under the post-TCRA condition relative to the baseline condition. A complete presentation of methods and results for the post-TCRA analysis, including the calculation of EPCs and the post-TCRA hazard characterization, is provided in Appendix F. The results of this evaluation are summarized briefly below.

Under the post-TCRA condition, the RME noncancer TEQ_{DF} HI is less than 1 for all hypothetical recreational fisher and recreational visitor scenarios evaluated. For the hypothetical subsistence fisher, only exposure scenarios that assumed consumption of catfish from FCA 2/3 in combination with direct contact to sediments had an RME TEQ_{DF} noncancer HI that exceeds 1 in the post-TCRA analysis. The RME noncancer TEQ_{DF} HI is 6 for these scenarios.

For all hypothetical scenarios (as well as for individual pathways) evaluated for the baseline risk assessment, the noncancer TEQ_{DF} HI was 3.3 fold higher than the cancer TEQ_{DF} HI. This is because the noncancer hazard and cancer hazard predictions used the same estimates of exposure and relied only on different toxicological criteria (i.e., the noncancer RfD of 0.7 mg/kg-day, and a cancer threshold TDI of 2.3 mg/kg-day). Under the post-TCRA condition, for all of the hypothetical recreational fisher and the recreational visitor scenarios evaluated, the cancer TEQ_{DF} HI is less than 1. For the hypothetical subsistence fisher, only exposure scenarios that assumed the consumption of catfish from FCA 2/3 in combination with direct contact to sediment has a post-TCRA RME cancer TEQ_{DF} HI greater than 1. The RME noncancer TEQ_{DF} HI is 2 for these scenarios under post-TCRA conditions.

The greatest reduction of hazards for both cancer and noncancer effects is for scenarios that assumed direct exposure to Beach Area E under baseline conditions. This is because the vast majority of TEQ_{DF} exposure and hazard for these scenarios is related to assumed direct contact rather than to the ingestion of fish or shellfish, and because exposure to sediment in this area is completely restricted under the post-TCRA condition. For these scenarios, the reduction in hazards related to the area under study resulting from the implementation of the TCRA range from 84 to 100 percent. For baseline exposure scenarios that assumed direct contact with sediments at Beach Area A, B/C, or D and the consumption of tissue from the adjacent FCA, the reduction in hazard ranges from 65 to 86 percent. A discussion of the uncertainties in this analysis is presented in Appendix F.

5.2.3.3 Probabilistic Risk Assessment

The results of the PRA provide insight into the variability of exposures and risks that may occur within the hypothetical potentially exposed population. Exposure and resulting noncancer hazards and dioxin cancer hazards for hypothetical young child fishers and young child recreational visitors were modeled in the PRA because this was the only age group for which HIs exceeded 1. The specific scenarios modeled are shown in Table 5-19. Only $COPC_{HS}$ defined as risk drivers were included in the PRA. These were TEQ_{DF} in sediment, tissues, and soils, PCBs in all tissue types, and methylmercury in catfish. The probability distributions used to model exposures in the PRA were discussed above in Section 5.1.2.2.1 and are presented in Appendix G.

Monte Carlo simulations were completed using Oracle® Crystal Ball software (Gentry et al. 2005). In order to investigate the numerical stability of the Monte Carlo calculations, 10 independent trials, each of 10,000 iterations, were run for two of the hypothetical receptor exposure scenarios being evaluated as part of the PRA (i.e., Scenarios 1A and 3A, chosen to represent one low-end and one high-end hazard scenario, respectively). The coefficients of variation[26] were 0.9–1.4 percent for the 50th percentile cancer and noncancer hazards and 1.6–2.4 percent for the 95th percentile cancer and noncancer hazards. On the basis of the relatively low variability indicated by these small coefficients of variation, 10,000 iterations were considered sufficient to produce stable numerical results.

For each of the hypothetical scenarios evaluated, the 50th, 90th, and 95th percentiles of the resulting output hazard distributions were summarized. The 50th percentile hazards represent estimates for individuals exposed under assumed average (or typical) conditions, while the 90th and 95th percentile hazards represent estimates for the individuals in the population assumed to be highly exposed. Table 5-22 presents the PRA results for noncancer

[26] The coefficient of variation is the standard deviation divided by the mean.

hazards and Table 5-23 presents the PRA results for dioxin cancer hazards. The results from the deterministic evaluations are included in these two tables for comparison.

5.2.3.3.1 Hypothetical Young Child Fisher

The model developed for each exposure scenario for the hypothetical young child fisher scenario included a range of exposures that was inclusive of the behaviors of both hypothetical recreational and subsistence fishing populations. The models were set up in this manner so that the impact of true variability in behaviors and patterns of exposure across the entire fisher population could be captured and explored. While the labels "recreational fisher" and "subsistence fisher" imply that there are two completely separate populations that have different and unique characteristics, it is appropriate to assume that there would be substantial overlap in the behaviors of average- and high-consuming individuals. For example, it is possible that some fishers assumed to consume large amounts of finfish on an annual basis only obtain a small portion of their total catch from within USEPA's Preliminary Site Perimeter, while other high consumers obtain most of their fish from this area. At the same time, there may be individuals assumed to consume at high rates but only fish within USEPA's Preliminary Site Perimeter during a single season while others fish there for many years. The same variations in behavior occur within the fisher population that consumes fish at more typical rates. Therefore, while some of the individuals modeled in the PRA may have behaviors that are similar to the behaviors modeled in the deterministic analysis for the hypothetical recreational fisher, and some may resemble the deterministic analysis for the hypothetical subsistence fisher, others will have characteristics that more closely resemble a combination of these populations. The PRA analysis for the hypothetical young child fisher was developed to capture the highly variable behaviors within the entire population of fishers. Details on the exposure probability distributions developed to represent the full range of potential fishing behaviors within a single model are provided in Appendix G.

When viewing the PRA results for the hypothetical young child fisher, the 50th percentile estimates represent hazards an individuals who may exhibit a combination of typical or average behaviors. The 90th and 95th percentiles characterize hazards for individuals who may participate in fishing activities that lead to high-end exposures.

As was seen in the deterministic evaluation, the PRA indicated that hypothetical scenarios that assumed direct contact with sediments at Beach Area E (in combination with the consumption of tissue from the adjacent FCA), exhibited significantly higher noncancer and dioxin cancer hazards than scenarios that assumed direct contact with sediments at Beach Area A, B/C, or D. The types of assumed exposures that contributed most significantly to the intake of $COPC_{HS}$ differed for these two subsets of scenarios. While direct contact pathways contributed the majority of the estimated exposure for scenarios that involved fishers at Beach Area E, the assumed consumption of finfish or shellfish was the most significant exposure route for all other scenarios evaluated. Therefore, in order to explore the impact of the variability in the exposure terms to overall intake and resulting hazards, these two subsets of fisher scenarios are discussed separately below.

Hypothetical fishers assumed to be exposed to sediments at Beach Area E, in combination with other exposures, were estimated to have the greatest noncancer and cancer hazards. The deterministic evaluation of these scenarios established that assumed exposure to dioxins and furans in sediment at this beach area contributed the vast majority (i.e., ≥98 percent in the deterministic evaluation) of the resulting hazards. Hypothetical fishers exposed to these sediments, and consuming catfish from FCA 2/3 (i.e., Scenario 3A) exhibited the highest overall hazard of any hypothetical fisher scenario evaluated. In the probabilistic analysis, none of the 50^{th} percentile endpoint-specific noncancer or cancer HIs for this hypothetical fisher population exceeded 1, however the 90^{th} and 95^{th} percentile HIs did exceed 1. The 50^{th}, 90^{th}, and 95^{th} percentile noncancer HIs for reproductive/developmental effects were 1, 8, and 10 respectively. For this same scenario, the 50^{th}, 90^{th}, and 95^{th} percentile HIs estimated for the immunotoxic endpoint were 0.4, 2, and 3 respectively (Table 5-22). These estimated reproductive/developmental hazards were attributable to potential exposure to TEQ_{DF} in sediment and catfish fillet, and methylmercury in catfish fillet. Estimated hazards for immunotoxicty were attributable to potential exposures to PCB in tissue. The 50^{th}, 90^{th}, and 95^{th} percentile TEQ_{DF} cancer HIs were 0.4, 2, and 4 (Table 5-23). For hypothetical fishers exposed to sediments at Beach Area E and assumed to consume clams or crabs, noncancer hazards for reproductive/developmental endpoints and dioxin cancer hazards were equal to those for the fisher described above. A sensitivity analysis revealed that the most influential factor in the overall variability in the noncancer and cancer hazards was the concentration of TEQ_{DF} in sediments, followed by the ingestion rate for fish or shellfish.

For the remaining subset of hypothetical fisher scenarios evaluated (i.e., those fishing scenarios that included assumed direct contact with sediments at Beach Area A, B/C, or D in combination with consumption of catfish or clam [Table 5-19]), consumption of tissue accounted for the majority of potential exposure to $COPC_{HS}$. For this subset of fishing scenarios, the 90th and 95th percentile dioxin cancer hazards were all below 1 (Table 5-23). Upper percentiles of endpoint-specific noncancer hazards for hypothetical fishers assumed to be exposed to sediments and consuming clams (i.e., Scenario 2B) were also below 1 (Table 5-22). For fishers hypothetically exposed to sediments outside of Beach Area E and consuming catfish the 50th, 90th, and 95th percentile estimates of HIs for developmental/reproductive effects were 0.5–0.6, 2, and 3–4, respectively (Table 5-22). For these same fishers, the 50th, 90th, and 95th percentile estimates of the HIs for the immunotoxic endpoint ranged were 0.4, 2, and 3, respectively (Table 5-22). The greatest sources of variability in the noncancer and cancer hazards for these scenarios were the assumed ingestion rate for fish and the fraction of fish ingested that were from within USEPA's Preliminary Site Perimeter. Collectively, these factors accounted for over 80 percent of the variability in predicted noncancer outcomes.

As discussed above in the deterministic evaluation of background risks and as demonstrated with the PRA using background concentrations (Table 5-22, Figures 5-8a and 5-8b), a portion of these estimated hazards were also present under exposure to background conditions. Figures 5-8a and 5-8b show the cumulative probability distribution for noncancer HI for reproductive/developmental effects for hypothetical fishers assumed to be exposed to sediments at Beach Area D and catfish fillet from FCA 1. Figure 5-8a shows the entire range of estimated hazards, and Figure 5-8b is truncated along the x-axis to provide greater detail around an HI of 1, and to more clearly illustrate the differences between risks within USEPA's Preliminary Site Perimeter and background for each percentile. These hazards were associated with potential exposures to TEQ_{DF} in sediment and TEQ_{DF} and methylmercury in catfish. The resulting cumulative probability distribution for hypothetical fishers assumed to be exposed to concentrations of TEQ_{DF} in background sediments and TEQ_{DF} and methylmercury in catfish fillet are also shown. For any given hazard percentile shown on Figures 5-8a and 5-8b, the horizontal distance between the two curves displays the incremental hazard assumed to be contributed by the area under study, i.e., the difference in hazards for the area under study relative to the hazard under background conditions.

With the exception of body weight and surface area, all exposure parameters were assumed to act independently and were not correlated in the PRA model. Bukowski et al. (1995) and Smith et al. (1992) reported that the effect of correlation between two variables on the output is most important if the correlation between the variables is high and if the correlation is a large contributor to the variance of the output. For the scenarios that include hypothetical exposures at Beach Area E and fish ingestion, the concentration of TEQ_{DF} in sediments and the fish ingestion rate were the largest contributors to overall variability in the predicted outcomes. For scenarios assuming direct contact with sediments in other beach areas and ingestion of fish or shellfish, the greatest sources of uncertainty in the noncancer and cancer hazards were the assumed ingestion rates and the fraction of fish ingested that were from within USEPA's Preliminary Site Perimeter.

It is possible that relationships may exist between some of the modeled variables (e.g., between body weight and ingestion rates); however, specific quantitative correlations for other sets of or pairs of exposure parameters (i.e., other than body weight and surface area) were not available. Moreover, it is not likely that strong correlations exist between the parameters that are known to drive variability in the predicted outcomes. Therefore not modeling these possible correlations is not likely to introduce a significant source of uncertainty into the PRA results.

5.2.3.3.2 Hypothetical Young Child Recreational Visitor

For the hypothetical young child recreational visitor, only a single scenario was evaluated using the PRA (Table 5-19). This scenario assumed a young child had direct contact (e.g., incidental ingestion and dermal contact) to both sediments at Beach Area E and soils throughout the area north of I-10 a few days a week for several years TEQ_{DF} was the only $COPC_H$ identified as a potential risk driver for soils and sediments, and therefore, only hazards associated with TEQ_{DF} were evaluated for the PRA.

For the hypothetical young child recreational visitor, estimated 50^{th}, 90^{th}, and 95^{th} percentile noncancer TEQ_{DF} HIs were 0.2, 2, and 4 respectively (Table 5-22). The estimated 50^{th}, 90^{th}, and 95^{th} percentile for cancer TEQ_{DF} HIs were lower at 0.05, 0.7, and 1, respectively (Table 5-23). The resulting probabilistic noncancer and cancer hazards associated with potential exposure to TEQ_{DF} in soils and sediments were more than an order of magnitude lower than the estimated deterministic TEQ_{DF} noncancer HI of 60 and cancer HI of 20.

5.2.3.3.3 Discussion of PRA Results

The results of the PRA provide insight into the variability of exposures and risks that may occur within the potentially exposed population. By comparing the deterministic estimates of hazards with the probability estimates, it is apparent that variability in various factors that influence exposure has a large impact on estimated hazards to the population (Tables 5-22 and 5-23). Because the deterministic RME estimates for the hypothetical young child did not account for these sources of variability, they likely overestimated any actual risks.

Even in the PRA, some aspects of variability were not accounted for. The probabilistic risk calculations were structured to use a single exposure point concentration for each iteration. This is equivalent to assuming that an individual eats fish containing the same $COPC_H$ concentration, or contacts soils or sediments with the same $COPC_H$ concentration, on every exposure event throughout his or her entire exposure period. In reality, it is more likely that hypothetically exposed individuals move around the area under study and are exposed to variable concentrations of $COPC_H$s over the durations of their assumed exposures. As a result, the exposure point concentrations to which they will actually be exposed will approach an average value over time. Such averaging would tend to pull both upper and lower tails of the risk distributions toward the central risk estimate, and would reduce estimates of upper percentile values. The impact of such an assumption on the model's output is largest when the actual variability in concentrations of a $COPC_H$ that a person could potentially contact is large. As exhibited by the sensitivity analysis for hypothetical fishers exposed to sediments at Beach Area E, this was the case for TEQ_{DF} in sediments at Beach Area E.

5.2.4 Uncertainty Analysis

According to USEPA (1989) guidance, risk characterization should also present information important to interpreting risks in order to place the risk estimates in proper perspective. There are numerous areas of uncertainty in any risk assessment, and assumptions made in the absence of information are often intentionally conservative and, therefore, tend to drive results toward overestimates of risk. Uncertainties exist in each step, including the data collection and analysis, the estimation of potential site exposures, and toxicity assessment. This section discusses the significant sources of uncertainty in this BHHRA.

5.2.4.1 Uncertainties in Data Collection, Analysis, and Treatment

There are a number of uncertainties related to data collection, analysis, and treatment. The more significant sources of uncertainty, as well as some that the EAM identified for discussion in this BHHRA, are discussed below.

In several samples from the 1966 perimeter of the northern impoundments, matrix interferences resulted in elevated detection limits for Aroclors. The use of these elevated detection limits for the sum of Aroclors would substantially overestimate sediment EPCs for total PCBs. Instead, one-half the detection limit for Aroclor 1254 in this subset of samples was substituted for deriving the EPCs for total PCBs. No Aroclors were detected in surface sediment within the 1966 perimeter and only a single detected concentration of Aroclor 1254 was measured at depth (2–4 feet) within this area (i.e., Station SJGB014, 1,400 µg/kg [qualifier J]). Moreover, in the *Screening Site Assessment Report* (TCEQ and USEPA 2006), which reported Aroclor results for several samples from within the wastes in the western cell of the northern impoundments, Aroclors were never detected. Aroclors were never detected in sediment samples, and detection limits for Aroclors in a number of sediment samples from within the northern impoundments were normal (9.5 µg/kg). In summary, there is uncertainty about the actual Aroclor concentrations in the materials collected from within the 1966 perimeter, but the estimated concentration of Aroclor 1254 at station SJGB014, and results of TCEQ and USEPA (2006) sampling, confirm that the approach taken to estimating total PCBs in sediment was conservative.

There are also uncertainties introduced with the data rules applied in the calculation of EPCs for the area under study. Following the data rules established for this assessment, TEQ_{DF} was calculated in two ways. First, individual congeners that were not detected in a sample were estimated to be present at one-half of the detection limit of that individual congener. Second, non-detected congeners were treated as zero. The impact of the decision on the resulting TEQ_{DF} is dependent on both the number of congeners that were not detected and the detection limits for the congeners that were not detected. By comparing the resulting EPCs calculated using these two approaches, the impact of the uncertainty was determined. The ratio of EPCs for TEQ_{DF} applying one-half the detection limit to TEQ_{DF} applying zero was generally small for the media and areas that resulted in the largest hazard. For sediments in Beach Area E and catfish from FCA 1 and FCA 2/3, the ratios were less than 1.05. Therefore, any uncertainty introduced by the treatment of non-detects did not substantially influence the risk results.

Consistent with comments received from USEPA on the Tissue SAP (Integral 2010b, Appendix C), total PCBs in tissue were evaluated as the sum of 43 specific PCB congeners (Table 3-3). This approach is consistent with that used by the Seafood and Aquatic Life Group (SALG) of the Texas Department of State Health Services (TDSHS) and is based on recommendations regarding the likelihood of occurrence in fish and the likelihood of significant toxicity (TDSHS 2008; MacFarland and Clarke 1989). Under the analytical methods used for measuring PCB concentrations in tissue, an additional 20 PCB congeners co-eluted with the 43 congeners of interest. These additional congeners, which included PCB-20, -30, -47, -61, -65, -69, -76, -83, -86, -90, -97, -109, -113, -115, -125, -129, -135, -163, -166, and -193, were also included in the sum for total PCBs. The use of this final metric for predicting hazards and risks from PCBs introduced some uncertainty into the risk assessment and may have resulted in overstated risks as the addition of these congeners, which are considered less toxic, means that the combined concentrations of the 43 specific congeners that are considered more toxic may have been overestimated. At the same time, there are other PCB congeners that were detected in sample results but were not included in this approach. The toxicities of these congeners are unknown but if any of these contribute additional toxicity to the mixture, then total risks to PCBs could be underestimated.

5.2.4.2 *Uncertainties in Exposure Estimates*

There are a number of uncertainties in the estimates of exposure at the area under study. These include both uncertainties regarding uses associated with the area under study, as well as the specific assumptions used to quantify risk. The more significant sources of uncertainty related to the exposure assessment are discussed below.

5.2.4.2.1 Minor Exposure Pathways

There are a number of minor exposure pathways for the area under study that were not evaluated quantitatively in this BHHRA. These included the potential inhalation of entrained dust derived from soil or sediment, inhalation of volatile compounds present in soil or sediment, and direct contact with surface water. While it is possible that these pathways could contribute additionally to total risk, any contribution would be very small, based on the $COPC_{HS}$ evaluated, and would not have affected estimated risks and hazards if they had been quantified.

Generally speaking, risks due to the inhalation of entrained dust originating from soils in the area under study are orders of magnitude lower than risks due to direct contact pathways (i.e., incidental ingestion and dermal contact) for soil. Therefore, any contribution from them is very minimal. In addition, because sediments have a high moisture content, it is not expected that they would provide a source of dust. While inhalation of volatiles in soil or sediment, if present, can contribute to total risk, none of the $COPC_{HS}$ identified is considered to be volatile.

It is possible that hypothetical receptors could be exposed to $COPC_{HS}$ in surface water, via incidental ingestion of surface water or via dermal contact during their activities, if those $COPC_{HS}$ are present in surface water. However, none of the $COPC_{HS}$ identified are likely to be dissolved in water at significant concentrations. The only other potential exposure routes to $COPC_{HS}$ in surface water would be dermal contact with or incidental ingestion of COPCs that are adhered to sediment particles suspended in the water column. Because direct contact with sediments (i.e., incidental ingestion and dermal contact) has already been evaluated for all hypothetical exposure scenarios, it is expected that these analyses are inclusive of any potential exposures that could occur through contact with surface water.

5.2.4.2.2 Hypothetical Trespassers

Potential exposures and associated risks were not quantified for a hypothetical trespasser exposed to media in the area north of I-10 and the aquatic environment. Although a hypothetical trespasser could be exposed via the same pathways as the hypothetical recreational visitor (i.e., direct contact pathways) and the hypothetical recreational fisher (i.e., ingestion of fish and shellfish), the hypothetical trespasser exposure would likely be intermittent and of a shorter term duration than the exposures assumed for either of those scenarios (e.g., chronic durations of up to 33 years). Therefore, for the area north of I-10, the estimated risks and hazards presented for the hypothetical fishers and recreational visitors overstate potential risks for the hypothetical trespassers.

Ingestion of catfish from the area under study and assumed direct contact with sediments at Beach Area E contributed to estimated potential noncancer and dioxin cancer hazards greater than 1 for hypothetical recreational fishers and recreational visitors. The highest potential noncancer and dioxin hazards associated with the ingestion of tissue for the hypothetical recreational fisher were 2 and 0.3 respectively. It is likely that any hypothetical trespasser would consume, on average, less than one-half the amount of tissue from the area under study as that assumed under the hypothetical recreational fisher scenario. Therefore the estimated noncancer and dioxin cancer hazards from ingestion of tissue would be less than 1 for the trespasser. Although the potential hazards assumed to occur with direct contact exposures at Beach Area E would likely be less for a hypothetical trespasser compared to the receptors evaluated quantitatively in this BHHRA, using the same model as employed for the quantitative risk assessment of other receptors, estimated noncancer and dioxin cancer hazards might be greater than 1 for a hypothetical trespasser with direct contact to sediments in this area.

Under post-TCRA conditions, it is possible that a hypothetical trespasser might have access to Beach Areas B/C and D. Any exposure to these areas would likely be intermittent. More frequent and longer term exposures assume to occur to sediments in these areas did not contribute significantly to risks for the hypothetical recreational fisher or visitor receptor groups evaluated. Therefore, potential direct contact exposure in these areas is also unlikely to contribute significantly to exposures and associated risks for a hypothetical trespasser.

5.2.4.2.3 Assumption of Age of Fishers and Recreational Fishers for CTE Estimates

In this BHHRA, the RME scenarios for all potential human receptor groups evaluated assumed that a portion of the total exposure occurs as a young child. This life stage represents a reasonable maximum because during this life stage, there is potential for higher exposures (on a per unit body weight basis) relative to other age groups. Inclusion of exposure parameters for small children in the RME scenarios is conservative, resulting in upper-bound exposure estimates for RME scenarios higher than any alternative age grouping. For the main CTE analysis, only adult exposures are evaluated, because it is hypothesized that adult individuals are the most likely to frequent the area under study.

It is recognized however, that adults may bring children with them under the adult scenarios evaluated. At the request of USEPA (comment 9 of the draft BHHRA, Appendix N) an additional CTE analysis was performed to evaluate potential CTE exposures for hypothetical young child fishers and visitors. This evaluation focused on estimating noncancer and TEQ_{DF} cancer hazard metrics because these metrics, unlike cancer risk estimates,[27] rely on an averaging time that is equal to the assumed exposure duration rather than averaging over a total lifetime, as is done in estimating cancer risk. Because of this, these hazard metrics are especially sensitive to life stages with higher relative intakes.

The CTE estimates used CTE EPCs (Tables 5-2 through 5-4) and relied on parameters that reflect central tendencies of behavior and exposure, relative to the upper-end estimates used to characterize RME risks. For parameters that describe the way the area of study is used (e.g., exposure frequency, fractional intake that is related to the area of study), the exposure parameter values assumed for the CTE adult analysis were adopted. Child-specific exposure parameters were developed from USEPA's (2011a) Exposure Factors Handbook, and the Lavaca Bay study (Alcoa 1998). Because the CTE parameters for the hypothetical young

[27] Cancer risks rely on a lifetime averaging time. For this metric the impact of a relatively short period where a receptor has a higher exposure relative to other periods does not have as large impact on the final risk estimates.

child fisher and visitor were not previously described in the EAM (Appendix A), a table listing the exposure parameter assumptions and their sources is provided (Table 5-24). A summary of the cumulative hazards for the CTE hypothetical young child fisher and visitor is provided in Table 5-25. The complete set of exposure and risk estimates for the CTE hypothetical young child fisher and visitor are presented in Appendix K.

Estimated CTE noncancer HIs and TEQ_{DF} cancer HIs for the hypothetical young child visitor were less than 1 for all scenarios evaluated. The noncancer HIs and TEQ_{DF} cancer HIs for the hypothetical young child fisher and visitor were two times the respective hazards estimated for the CTE adult.

5.2.4.2.4 The Presence of Subsistence Fishers

The hypothetical subsistence fisher scenario was evaluated to address the concern raised by USEPA that there might be individuals who fish exclusively from within USEPA's Preliminary Site Perimeter over an extended period of time to provide food for themselves and other family members and, therefore, consume more fish from the area than other recreational anglers. While the Public Health Assessment for the Site (TDSHS 2012) describes the northern impoundments as having once been a popular fishing location, the ancillary evidence neither supports the presence of a subsistence fishing population, nor does it support the conclusion or assumption that the area has been heavily and consistently used by the same individuals for fishing at a subsistence level across decades.

While there may be individuals who are high level consumers within any angler population that uses a particular fishery, high level consumption can rarely be predicted based on socioeconomic characteristics such as income level and ethnic, racial or cultural background. As discussed in Appendix L, it is rare that true subsistence fishing populations are found. The use of the word "subsistence" can be taken to mean that the individual is living, in whole or in part, at the minimum level of food/and or shelter needed to support life. In the context of fishing, however, it typically refers more generally to an individual who relies on self-caught fish as a primary source of dietary protein. Among various subpopulations, cultural, ethnic, or socioeconomic factors may influence fish consumption habits and behaviors. For these reasons, the potential subpopulations that might have subsistence ingestion rates include: 1) low income individuals who depend on self-caught fish to supplement their diets, and/or 2) ethnic groups (such as some Native American tribes) for which consumption of substantial quantities of fish has historically been part of their cultural tradition.

Given the general lack of predictability of subsistence behaviors based on demographic characteristics, and the very low likelihood that long-term subsistence fishing is occurring within USEPA's Preliminary Site Perimeter (TDSHS 2012), the subsistence fisher, as evaluated in this BHHRA, is hypothetical and unlikely to have been present or to be present in the future in the area under study. This is made even more unlikely under current (post-TCRA) conditions because any access to the area within USEPA's Preliminary Site Perimeter is highly restricted by fencing.

5.2.4.2.5 Estimated Exposure from Fish Consumption

A number of the assumptions used in estimating exposure to $COPC_{HS}$ in finfish and shellfish are uncertain. These include the selection of one tissue type to represent all types of fish that an individual may consume, the selected finfish and shellfish ingestion rates assumed, and the chemical reduction due to preparation and cooking.

Use of Hardhead Catfish as a Conservative Representation of Ingested Fish

In this BHHRA, exposures associated with assumed finfish consumption were estimated using catfish fillet data. Although no information about species preferences or catch rates from within USEPA's Preliminary Site Perimeter is available, it is unlikely that any individuals who fish and who consume fish from the area under study would consume only catfish at the ingestion rates assumed. It is more likely that they would consume a mixed diet that includes a variety of fish types. In the Lavaca Bay study, only one individual of the 1,751 anglers who reported fish consumption in that survey reported that he and his family consumed hardhead catfish during the month-long study period. Even when all types of catfish that were reported (hardhead, gafftopsail, blue, and channel catfish) were combined, only 148 (less than 1 percent) of 15,778 meals reported were catfish of any species.

Hardhead catfish are benthic fish, which tend to accumulate higher concentrations of dioxins and furans than pelagic fish (USEPA 2009a), although pelagic fish are more generally targeted for consumption. On the basis of data presented in the Tissue SAP, the selection of hardhead catfish to support the risk evaluation provides a conservative representation of edible fish tissue. In preparation of the Tissue SAP, available tissue chemistry data for hardhead catfish, blue crab, and blue catfish collected from within USEPA's Preliminary Site Perimeter were evaluated. For the two catfish species, the mean, minimum, and maximum TEQ_{DF} concentrations were higher in hardhead catfish fillet from within USEPA's Preliminary Site Perimeter than in blue catfish fillet from the same area. A review of available Category 2 data collected in 2005 and later from the regional area (outside of USEPA's Preliminary Site Perimeter) shows that median TEQ_{DF} concentrations in hardhead catfish are also higher than median concentrations in the fillets of other species including blue catfish (*Ictalus furcatus*), striped bass (*Morone saxatilis*), red drum (*Sciaenops ocellatus*), smallmouth buffalo (*lctiobus bubalus)*, southern flounder (*Paralichthys lethostigma*), and spotted seatrout (*Cynoscion nebulosus*). Maximum concentrations of TEQ_{DF} in regional hardhead catfish are higher than maximum concentrations in regional striped bass, red drum, smallmouth buffalo, southern flounder, and spotted seatrout (Figure 5-9) (TDSHS 2010; University of Houston and Parsons 2006). These data suggest that if a mixed diet of various fish types was modeled for this BHHRA, the resulting hazards (both cancer and noncancer) from TEQ_{DF} would be lower than were estimated here. The precise difference in that risk is unknown and would vary, depending on the species mix considered.

The absence of information on ages of fish analyzed and age-preferences of anglers is a source of uncertainty in the risk assessment. This is because mercury and PCBs have the potential to increase in concentration with fish age. In the case that the ages of fish analyzed do not correspond with those that are fished and ingested by anglers within USEPA's Preliminary Site Perimeter, exposure to $COPC_{HS}$ may be under- or overestimated by the exposure evaluation. Research evaluating the relationships between fish age and tissue concentrations of mercury, PCBs and dioxins is discussed briefly below.

Some research has shown that methylmercury can accumulate in fish tissue over time, resulting in a correlation between fish age and mercury concentrations in tissue (e.g., Lange et al. 1993; Grieb et al. 1990). It appears that there may be some potential for PCB concentrations to increase with fish age, but there is no support for a general assumption that concentrations of dioxins and furans increase with fish age. There is evidence to suggest that most dioxins and furans do not increase with fish age, and in fact concentrations of some congeners decrease with age (Wang and Lee 2010). The absence of age-related increases in concentrations of dioxins and furans in fish tissue is consistent with the findings of the *Technical Memorandum on Bioaccumulation Modeling* (Integral 2010c). That document synthesizes various sources of information and concludes that dioxins and furans have limited potential for bioaccumulation and biomagnification in fish and benthic invertebrates because there are biological limits on uptake and because fish and invertebrates can metabolize and excrete dioxins and furans to an extent that varies for the different congeners.

In light of this information, it is possible that if smaller or younger fish than those ingested by anglers were sampled during the fish study, mercury and possibly PCB exposures may have been underestimated. Given that fish age information was not collected as part of the tissue study, and given the lack of any quantitative understanding on size-concentration relationships, it is not possible to predict the degree of such an underestimation, if one was expected to exist. However, the size range targeted for collection in the tissue study was at the upper end of the range of sizes observed among all hardhead catfish collected by Yanez-Arancibia and Lara-Dominguez (1988) in the Gulf of Mexico. Although the size distribution in the population of hardhead catfish that may occur within USEPA's Preliminary Site Perimeter is unknown, the information on sizes of this species in the Gulf of Mexico suggests that the sizes of hardhead catfish targeted and captured in the tissue study conducted for the RI were the largest among the hardhead catfish that could occur within USEPA's Preliminary Site Perimeter. It should be noted that the relative concentrations between the exposures estimated for the area within USEPA's Preliminary Site Perimeter and background would remain constant. Hazards associated with background exposure to methylmercury in catfish fillets were similar to or higher than hazards within USEPA's Preliminary Site Perimeter, indicating that any exposures from the study area are not contributing additional risks due to methylmercury.

Regardless of age/size preferences of anglers, there is no basis for concern that the fish collected could have resulted in a downward bias in the exposure assessment for dioxins and furans. In fact, the tissue data used in the risk evaluation likely resulted in an upward bias in the human exposure assessment because hardhead catfish are a benthic fish (USEPA 2009a), and have been demonstrated to have higher TEQ_{DF} concentrations than other species captured both within and outside of USEPA's Preliminary Site Perimeter.

Shellfish and Fish Consumption Rates

Although the fish and shellfish ingestion rates from Lavaca Bay were determined to be the best available for this BHHRA, there is some uncertainty with their application for this BHHRA. As part of this BHHRA, in addition to the reported results, the raw data for the Lavaca Bay study were reviewed and provided additional insight into some of the uncertainty associated with these rates. For the young child, the data included 326 records for children who consumed finfish during the study period. However, during that same period, only 29 of these child consumers were reported consume shellfish despite the fact that they were fish consumers. Consequently, the population of fish consumers was quite large, but the subset of individuals who consumed shellfish was quite small. Similar differences are observed if other types of fish are segregated.

The report on the Lavaca Bay study handled this by including zero values for all of the fish consumers who did not consume shellfish during the study period in deriving the reported statistics for consumption rates for shellfish. The inclusion of these zero values resulted in central tendency and upper-bound consumption rates that were lower than they would have been if only those 29 children who consumed shellfish were considered in estimated consumption rates. An alternative approach would have been to develop a distribution that was based only on the consumption rates reported for individuals who actually consumed shellfish (i.e., 29 children). Using this approach, the median value was 4 g/day and the 95^{th} percentile was 13 g/day. Had these values been applied as the CTE and RME ingestion rates for the young child, the resulting risks and hazards associated with the shellfish consumption pathway would have been approximately 7-fold higher than those presented for this BHHRA.

Fraction of Ingested Fish from within USEPA's Preliminary Site Perimeter

There is also uncertainty regarding the amount of fish that individuals eat that are from within USEPA's Preliminary Site Perimeter. Information on the fractional intake of fish and shellfish from various areas as reported in the Lavaca Bay study were used to inform the value assumed for this parameter in this BHHRA. Alcoa (1998) reported that the mean and

95UCL fractional intakes of finfish in the 1,500 acre closure area studied within Lavaca Bay were less than 10 percent, and the fraction of shellfish consumed from the area was even lower, at less than 1 percent. For this BHHRA, RME and CTE fractional intakes for fish and shellfish of 0.25 and 0.1 were assumed for the hypothetical recreational fisher scenario, and a fractional intake of 1.0 was assumed for the hypothetical subsistence fisher scenario. The assumed values are likely conservative; however, the lack of information specific to the area under study does not allow the term to be more accurately defined for this BHHRA.

Cooking Loss

Another uncertainty in estimating exposure from the ingestion of tissue is related to the loss factor assumed for preparation and cooking. It is well recognized that tissue preparation and cooking methods used may reduce chemical concentrations in fish tissues, particularly for lipophilic compounds such as dioxins, furans, and PCBs (USEPA 2000b, 2002b; Wilson et al. 1998). Because these compounds are lipid soluble, removing the fat prior to consuming the fish will reduce dioxin exposure. U.S. Food and Drug Administration (FDA 2013) and various fishing advisories recommend that food preparation methods that remove fat, such as trimming fat and cooking fish, be used to reduce potential exposures to dioxins and PCBs in tissue. Information on the specific cooking techniques that remove the largest amount of dioxins, furans, and PCBs is not known, however it is thought that those that allow for the fat to be separated or drained from the tissue are likely to reduce exposure the most. There is some uncertainty, however, regarding the precise amount of chemical-specific reduction that occurs. For the deterministic exposure evaluation, a cooking loss term of 0 percent (no loss) was conservatively assumed for PCBs and dioxins.

As established in the EAM (Appendix A), the impact of assuming a cooking loss factor of 0.25 (25 percent) was explored in the uncertainty evaluation for this BHHRA. In addition, the PRA applied distributions for this chemical reduction factor for dioxins and PCBs. These distributions are described in detail in Appendix G.

The loss parameters were applied to catfish fillets only, and not to clams or crabs. TEQ_{DF} and PCBs contributed a substantial amount of the potential noncancer hazards from catfish ingestion. There is a direct linear relationship between the cooking loss factor for a chemical and total intake of (and hazard or risk attributable to) that chemical from tissue. Therefore, when a cooking loss factor of 0.25 was applied, the noncancer hazards and risks attributable to TEQ_{DF} and PCBs were reduced by 25 percent.

For the hypothetical recreational fisher, when cooking loss was assumed to be zero, the assumed consumption of catfish tissue from FCA 2/3 resulted in a noncancer HI of 2.3 for all $COPC_{H}$s and an HI of 2.0 for TEQ_{DF} and total PCBs combined. Applying a loss factor of 0.25 resulted in reduced HIs of 1.8 and 1.5, respectively, or a 21 percent reduction in total hazard. The contribution of TEQ_{DF} and PCBs to overall hazard from consumption of catfish was similar for FCA 1 (i.e., 85 percent compared to 83 percent). Applying the cooking loss factor of 0.25 resulted in a 21 percent reduction in total hazard attributable to consumption of catfish in FCA 1. The relative impact of this factor (i.e., 21 percent) on the resulting noncancer hazards for the hypothetical subsistence fisher was the same as for the hypothetical recreational fisher.

No loss factors were evaluated for clams or crabs because no data on chemical reduction due to preparation and cooking specific to shellfish could be located. Clam tissue analyzed from locations within USEPA's Preliminary Site Perimeter had a substantially lower percent lipid than most finfish, and techniques used for preparing and cooking shellfish differ from those used for finfish. Therefore, no alternative cooking loss factor was explored for shellfish. However, if there is also a loss of $COPC_{H}$ concentrations when shellfish are cooked, then the estimated risks and hazards may be over-stated.

A recent meta-analysis published by AECOM (2012) reviewed the available data on cooking loss for lipophilic compounds. Studies completed in a variety of tissue types and applying a range of preparation and cooking methods were reviewed, and those with sufficient data for quantitative analysis were used to determine the range and midpoint of cooking loss for dioxins and PCBs. The analysis focused on studies that used a relevant and appropriate experimental method and presented changes in raw and cooked fish tissue COPC levels on a mass basis because a comparison of concentrations in raw and cooked fish alone neglects the change in tissue mass that occurs with cooking, which is often significant. The median losses were generally in the range of 20 to 50 percent for typical cooking methods and consistent differences in mass loss between cooking methods were not apparent. Across all tissue types and cooking methods, the median losses were 32 percent for PCBs and 50 percent for dioxins and furans. The results of this recent meta-analysis suggest that the hazards presented in the deterministic risk assessment, which relied on a loss of 0, are conservative, and that the impact of actual losses is even greater than those discussed above in this uncertainty evaluation which assumed a 25 percent loss factor.

5.2.4.2.6 Estimated Exposure from Direct Contact Pathways

There are also some uncertainties associated with certain assumptions used for estimating exposure via direct contact. These include the use of a maximum concentration of dioxins and furans for the EPC at Beach Area E, adopted sediment adherence factors and assumptions about exposure patterns and frequencies.

Employing the rules established in the EAM for selecting EPCs, the maximum concentration of TEQ_{DF} in sediments at Beach Area E was selected as the EPC for the RME estimate (Appendix E). The selection of this maximum concentration introduced a large amount of uncertainty into the risk estimates for direct contact with sediments at this Beach Area. At Beach area E, TEQ_{DF} concentrations ranged from 8.5 to 13,000 ng/kg and the geometric mean concentration was 910 ng/kg. The RME EPC of 13,000 essentially resulted in the assumption that individuals are exposed to only sediment with this high concentration of TEQ_{DF} for the entirety of time spent in this area. It is much more likely, that over an extended duration, under the baseline condition considered in this BHHRA (i.e., immediately prior to the TCRA), individuals would have been exposed to an average concentration of dioxins and furans present in sediments at this Beach Area. The geometric mean concentration of 910 ng/kg was adopted as the EPC for the CTE estimates. The CTE cancer and noncancer TEQ_{DF} HIs for direct contact with sediments at this Beach Area were 0.08 and 0.3 respectively, and were two orders of magnitude lower than the corresponding RME estimates. Although the differences in the CTE and RME estimates reflected other differences in the scenarios in addition to the EPC, the 14-fold difference between the RME and CTE EPCs assumed for this scenario was one of the factors that heavily influenced the differences in these estimates. Given the wide range of variability in TEQ_{DF} concentrations in sediments present at Beach Area E, as described above this assumption had large implications for the risk results. Specifically, TEQ_{DF} noncancer and cancer hazards are 14-fold lower when the CTE EPC is assumed in place of the maximum concentration.

Few studies have evaluated adherence of sediments to exposed skin; however, it has been established that adherence for wet soil or sediment are generally higher than for dry soil (USEPA 2011a; Bergstrom et al. 2011). In addition to the moisture content of the exposure medium, the particle size makeup of the medium may impact adherence. The sediment values presented in USEPA (2011a) and used for the deterministic evaluation were based on body part-specific adherence factors from Shoaf et al. (2005). This study measured sediment adherence in children playing in tidal flats composed primarily of sandy sediments, and established adherence factors ranging from 0.042 mg/cm^2 for the face to 21 mg/cm^2 for the feet. These body-specific adherence factors were used to determine a weighted adherence factor of 3.6 mg/cm^2 for the hypothetical young child fisher and recreational visitor scenarios.

The sediments at Beach Areas A, B/C, D, and E consist of a range of particles with the bulk being finer grained sediments including silt, very fine sand, and fine sand (Figure 5-10). Overall, these sediments appear to be finer than those studied by Shoaf et al. (2005) and, therefore, there is a degree of uncertainty introduced by the use of these factors in this BHHRA. Given the higher concentrations of $COPC_{HS}$ in sediments at Beach Area E, the impact of this uncertainty is greatest for the hazards and risks estimated for direct contact with Beach Area E. For example, if an adherence factor for soil had been applied in the place of that for sediments, hazards resulting from direct skin contact with sediment would have been reduced by more than an order of magnitude.

The theoretical relationship between particle size and the mass required to provide monolayer coverage is important to understanding the potential for chemical absorption. Monolayer loading is defined as the complete coverage of skin with one layer of particles. Experimental results show that the monolayer is a critical level: soil layers above the monolayer contribute very little to dermal absorption (USEPA 2011a). The soil load required to reach a monolayer depends on the particle size of the soil. Using the relationship established by Duff and Kissel (1996), the load representing monolayer ranges from 4.3 mg/cm^2 for clay particles to 208 mg/cm^2 for course-grained sand. This theoretical demonstration is a simplification for any real application because real layers of soil or sediment consist of heterogeneously sized, irregular particles, however the large resulting range in monolayer loads demonstrates the large amount of potential variation in true adherence.

5.2.4.3 Uncertainties in Toxicity Evaluation

Dioxins and furans were defined as risk-driving chemicals in sediments, soils, and tissue. PCBs were defined as risk-driving chemicals in tissue. Therefore, the focus on the uncertainties introduced by the toxicological criteria applied for this BHHRA are focused around those $COPC_{HS}$. While mercury was also defined as a risk driving chemical in catfish, the mercury concentrations in catfish were not statistically different from background mercury concentrations in catfish. Therefore, uncertainty in the toxicological evaluation of mercury is not further discussed.

5.2.4.3.1 Dioxin and Furan Toxicity

The toxicity criterion that was used to evaluate potential cancer risks due to dioxins and furans (i.e., as TEQ_{DF}) was the TDI of 2.3 pg/kg-day derived from JECFA (2002). This TDI was developed based on the assumption that the cancer dose response for TCDD and other DLCs is not linear and that there is a threshold for the carcinogenic effects of these compounds. There is substantial support for using a threshold approach to evaluate DLCs (WHO 1991, 1992, 1998; JECFA 2002; Simon et al. 2009; NAS 2006; ACC 2010; TCEQ 2010a,b, 2011; Haney 2010).

While the threshold-based approach for carcinogenic effects has been discussed in the draft dioxin reassessment, it has not yet been adopted by USEPA as the basis for its cancer-based toxicological criterion. USEPA's historical approach has been to assume that the carcinogenic effects of dioxins and furans have no threshold dose, and to use a CSF to evaluate potential cancer risks, assuming that the dose response is linear. As discussed in Section 4.3.1, USEPA has been conducting its dioxin reassessment for nearly 20 years. While the scientific consensus during that period has been growing to conclude that DLCs act via a non-linear dose response, USEPA's most recent report on its reassessment indicates that it has not yet changed its assumption that TCDD acts as a non-threshold carcinogen.

Historically, USEPA has used an upper-bound CSF of 150,000 (mg/kg-day)$^{-1}$ for TCDD (USEPA 1997b), based on the increased incidence of hepatocellular and respiratory tumors reported in the Kociba et al. (1978) study and extrapolation using a linearized multistage model.[28] It should be noted, however, that in addition to the value that was developed by USEPA using these data, a number of other agencies and independent scientists have used the same data to derive a variety of linear-based CSFs for TCDD. These CSF estimates have ranged from 9,000 to 156,000 (mg/kg-day)$^{-1}$ (USEPA 1985, 2000a; FDA 1993, 1994; Keenan et al. 1991). The differences among them are the result of changes in tumor classification protocols that have occurred since the earlier studies were conducted, selection of approaches for scaling from animals to humans, early mortality corrections, the selected tumor types upon which the dose response models are based, and the choice of the specific linear extrapolation model used to evaluate them. Therefore, the decisions that must be made in extrapolating the results from animal studies to derive a CSF can greatly impact the resulting CSF estimates, adding greatly to their uncertainties, even when the same starting data are used.

Further uncertainty in the CSF approach is introduced considering that other scientists have developed CSFs based on data that are more recent than the Kociba et al. (1978) study. California EPA (CalEPA 1986) completed multiple analyses and based its CSF of 130,000 (mg/kg-day)$^{-1}$ on the incidence of liver tumors in male mice observed in the National Toxicology Program (NTP) mouse bioassay (NTP 1982). Subsequently, the California OEHHA (2007) used a CSF of 26,000 (mg/kg-day)$^{-1}$, which was based on the results of a more recent NTP (2006) study, in deriving its 2007 drinking water criteria. Simon et al. (2009) developed a CSF of 100,000 (mg/kg-day)$^{-1}$ using the same NTP (2006) dataset but used a body burden approach, rather than an administered dose, to derive a linearized CSF. Finally, USEPA (2011b) has indicated that it may increase its CSF to 1,000,000 (mg/kg-day)$^{-1}$, based on its application of a linear dose response approach model to epidemiological data.

Alongside the wide range in estimated CSF values that assume a linear dose-response relationship between TCDD and cancer, there is growing worldwide consensus that TCDD's cancer effects have a threshold. A number of agencies and scientists have derived

[28] USEPA (1985) published a slightly higher CSF of 156,000 (mg/kg-day)$^{-1}$ in its 1985 Health Assessment document based on these same data.

toxicological criteria that are based on a threshold dose instead of a linear dose-response model. These toxicological criteria range from 1 to 100 pg/kg-day. Simon et al. (2009) derived an RfD of 100 mg/kg-day for the cancer endpoint using the 2006 NTP data. The World Health Organization (WHO) (1991, 1992) developed a TDI of 10 pg/kg-day, which it believed to be protective of cancer effects, based on its review of the available toxicological literature. Subsequently, in concert with the International Programme on Chemical Safety, WHO developed a revised TDI range of 1 to 4 pg/kg-day, based on body burden data and using a steady state pharmacokinetic model, that it considered protective of both cancer and noncancer endpoints. In addition, JECFA's recommended toxicological criterion, which provides the basis for the TDI of 2.3 pg/kg-day that is used in this BHHRA, was based on body burdens reported for two animal studies. Table 5-26 provides a summary of key toxicological criteria that have been developed for TCDD.

The CSFs that have been derived using linear dose response modeling are not directly comparable to the dose-based toxicological criteria that have been developed, assuming that there is a threshold. It is possible, however, to compare the risk-specific doses[29] (RsDs) that can be derived using the CSFs with the threshold-based values.

Using a target cancer risk level of 1×10^{-4} to convert the various upper-bound CSFs ranging from 9,000 to 156,000 $(mg/kg\text{-}day)^{-1}$ to RsDs results in RsDs ranging from 0.64 to 11 pg/kg-day. The target risk of 1×10^{-4} was selected as the basis for this comparison because it is the upper-bound of USEPA's target range for incremental cancer risk. The values that have been derived assuming that DLCs act as threshold carcinogens range from 1 to 100 pg/kg-day. The JECFA value that was used in this BHHRA is higher than the lowest RsD by a factor of 3.6, but is lower than the upper end of that range by roughly a factor of 5. It is also at the low end of the range of threshold-based criteria; 2.3 times higher than the lowest value reported but more than 40 times lower than the upper end of that range. This indicates that while the TDI of 2.3 pg/kg-day is not the most conservative value that could have been used, it is well within the range and near the low end of the toxicological criteria that have been used by other agencies worldwide.

[29] A risk-specific dose is the dose level that is associated with a specified level of cancer risk. It is calculated by dividing a target risk level by the chemical-specific CSF to determine the chemical-specific dose level that results in that cancer risk.

Although USEPA has not finalized its dioxin reassessment, its 2003 draft proposed a linear-based CSF of 1,000,000 $(mg/kg\text{-}day)^{-1}$. When this CSF is used to develop an RsD based on a 1×10^{-4} target risk, it results in an RsD of 0.1 pg/kg-day. This is lower by nearly a factor of 7 than the lowest of the RsDs derived from Tier 3 studies (Table 5-26). The JECFA value is higher than that value by a factor of 23.

To meet requirements articulated by USEPA in comment 1 (Appendix N) on the draft of this document, a sensitivity analysis of TEQ_{DF} cancer hazards and TEQ_{DF} cancer risks was completed. Tables 5-27 through 5-31 report TEQ_{DF} cancer hazards calculated for the area of study using the TDI of 2.3 pg/kg-day and TEQ_{DF} cancer risks with the CSF of 156,000 $(pg/kg\text{-}day)^{-1}$. Tables 5-32 and 5-33 present TEQ_{DF} cancer hazards and cancer risks for background. To convey the cumulative impact of estimating TEQ_{DF} cancer risk with the CSF approach, cumulative cancer risks for other carcinogenic $COPC_{HS}$ and TEQ_{DF} are also shown. Scenarios that resulted in RME TEQ_{DF} cancer HIs greater than 1 also resulted in TEQ_{DF} cancer risks greater than 1×10^{-4} (Tables 5-27, 5-29, and 5-30). The greatest estimated hazards and risks were for hypothetical recreational fishers, hypothetical subsistence fishers, and hypothetical recreational visitors with assumed exposure to sediments at Beach Area E. No other scenarios resulted in a TEQ_{DF} cancer HI greater than 1 or TEQ_{DF} cancer risk greater than 1×10^{-4}; however, estimated cumulative cancer risks for all other RME scenarios were greater than 1×10^{-6} (Tables 5-27, 5-29, and 5-30). The estimated RME cumulative excess cancer risk for the hypothetical subsistence fisher exposed to background sediment and ingesting catfish was 2×10^{-4}, and also exceeded the upper end of the range of excess cancer risk considered in management decisions by USEPA (Table 5-32). As was the case for RME risks estimated for the area under study, cumulative excess risks for all RME scenarios in background exceeded 1×10^{-6}. Cumulative cancer risks for CTE scenarios ranged from 3×10^{-8} to 7×10^{-6} and 3×10^{-8} to 9×10^{-6} for adult recreational fishers, and adult recreational visitors respectively (Tables 5-28 and 5-29). Although USEPA has not established a CSF for assessment of dioxin cancer risk, there is substantial technical support for the use of the TDI instead of the CSF in risk assessment (Appendix B of Integral 2012b).

There are also substantial uncertainties associated with USEPA's recently published RfD of 0.7 pg/kg-day for TCDD, which was used to evaluate the noncancer effects of DLCs in this

BHHRA. This value was based on studies conducted by Baccarelli et al. (2008) and Mocarelli et al. (2008). Both evaluated health effects in human populations that were exposed to dioxins and furans as the result of a trichlorophenol reactor accident that occurred in 1976 in Seveso, Italy (USEPA 2012c).

While this RfD has been adopted by USEPA, a number of questions arose during its peer review pertaining to the selection of appropriate NOAELs, pharmacokinetic consideration of increased elimination rates in children, correction for exposures to other dioxins and furans, and the full weight of evidence provided by other human and animal studies (SAB 2011; ACC 2010; Foster et al. 2010). USEPA did not resolve all of those issues prior to publishing the value in its IRIS database.

Differing values for noncancer effects have also been developed by other agencies worldwide. The ATSDR, WHO, the Joint United Nations Food and Agriculture Organization, the European Commission Scientific Committee on Foods, the Japanese Ministry of Health and Welfare, and the Health Council of the Netherlands and JECFA all derived dose-based quantitative health guidelines ranged from 1 to 4 pg/kg-day based on a number of different, noncancer, toxicological endpoints for TCDD and DLCs (DeRosa et al. 1999; Pohl et al. 2002; JECFA 2002). The lower end of that range is roughly 50 percent higher than USEPA's RfD and the upper end of that range is higher by nearly a factor of 6. Given the uncertainty in the actual noncancer toxicity of DLCs it is possible that the use of USEPA's RfD to evaluate noncancer hazards may have overestimated those hazards by as much as a factor of 6.

A substantial amount of the potential risks and hazards for the area under study were associated with potential exposures to DLCs in sediments and fish/shellfish tissues. Using the dioxin cancer hazard approach results in an estimated cancer hazard for the mixtures of these compounds measured in these media. Like a noncancer hazard index, if the cancer hazard exceeds 1, USEPA assumes that there is a potential for developing cancer within the exposed population based on exposure over the assumed exposure duration, while if the cancer hazard does not exceed 1, it is concluded that there is no risk of developing cancer. This differs from USEPA's traditional approach of estimating an incremental increase in potential cancer risk for carcinogenic compounds and comparing that risk to USEPA's target risk range of 1×10^{-4} to 1×10^{-6}.

The result of this is that reported cancer hazards for DLCs are not directly comparable, and, therefore, cannot be summed with the incremental cancer risks reported for the other carcinogenic compounds. While this can appear to complicate the interpretation of risk results, it is appropriate not to sum them. This is because the calculated cancer hazard, using the TDI, is similar to the endpoint-specific noncancer hazard. Therefore, if the cancer hazard exceeds 1, using USEPA's thresholds, USEPA assumes that there may be some risk of cancer under the assumed hypothetical scenario, whereas if it does not exceed 1, then it is assumed that the DLCs do not contribute to the potential cancer risk for that same scenario.

5.2.4.3.2 PCB Toxicity

As discussed in the TESM (Appendix B) there is some uncertainty associated with the way in which PCBs were evaluated. USEPA's IRIS database, which presents CSFs and RfDs for PCB mixtures with variable degrees of chlorination, also states that (USEPA 2011a):

> "when congener concentrations are available, the slope-factor approach can be supplemented by analysis of dioxin TEQs to evaluate dioxin-like toxicity. Cancer risks from dioxin-like PCB congeners (evaluated using dioxin TEQs) would be added to risks from the rest of the mixture (evaluated using slope factors applied to total PCBs reduced by the amount of dioxin-like congeners)."

While both of these approaches contribute uncertainties to the estimation of risks and hazards due to PCB, the uncertainties associated with the use of toxicological criteria that USEPA has developed for PCBs contributes less uncertainty.

USEPA's CSF for highly chlorinated PCB mixtures, which was used in this BHHRA, is based on upper-bound estimates of the toxicity of Aroclors 1248, 1254, and 1260. The RfD for Aroclor 1254 was used for evaluating noncancer hazards from potential exposure to PCBs. As long as the congener mixtures present in the exposure media are similar to these Aroclors, the risk and hazard estimates based on these criteria should be reliable and conservative. This is because the observed toxicity upon which the criteria have been based, represents the combined toxicity associated with all congeners that are present in that mixture. The observed toxicity, therefore, accounts for the contributions of all of the components of the mixture, their potential additivities, their agonistic and antagonistic interactions, and their competition for the same binding sites.

It is acknowledged, however, that congener mixtures in environmental media may differ from the Aroclor mixtures due to variations in congener uptake and bioaccumulation, and losses or alterations in the mixture due to weathering. This is one of the reasons that USEPA recommends using the TEQ approach to evaluate PCBs in addition to the PCB-specific toxicological criteria. There is concern that the composition of the PCB mixture that is present in media in the area under study may differ from the PCB mixture used to derive the toxicological criteria, due to aging and the variable physical/chemical properties of the different congeners, so that the mixture no longer resembles the mixture upon which those criteria are based. Depending upon the congeners present, the toxicity of the aged congener mixture could be greater or less than the upper-bound values presented in IRIS.

To evaluate this possibility, an analysis of the PCB congener composition in the tissue used in this BHHRA was completed to determine whether it resembled the highly chlorinated mixtures upon which USEPA's recommended CSF and RfD are based. Specifically, the percent congener composition of Aroclors 1248, 1254, and 1260, as reported by Newman et al. (1998) were compared with the percent composition of congeners measured in the biota to determine whether the weathering and differential uptake may have resulted in a congener mixture in biota tissue that did not resemble that of the highly chlorinated Aroclor mixtures upon which USEPA's toxicological criteria are based.

As shown in Figures 5-11 through 5-13, that analysis indicated that the congeners present in catfish, clams, and crabs most closely resembled Aroclor 1254 or a mixture of Aroclor 1254 and 1260 and so also resembled those mixtures upon which the USEPA's toxicological criteria were based. Therefore, it can be concluded that the estimated risks and hazards for ingestion of PCBs in biota were appropriate and conservative estimates.

The alternative approach of evaluating TEQp, as presented by USEPA (2012c), contributed greater uncertainty to risk and hazard estimates for PCBs for a number of reasons, which are discussed below.

USEPA recommends evaluating the 12 dioxin-like PCB congeners using the toxicological criteria for TCDD, subtracting out their concentrations from the concentration in the total PCB mixture, and then evaluating the remaining mixture of 197 congeners using the toxicological criteria that were specifically developed for PCB mixtures. The health effects upon which USEPA has derived its toxicological criteria for PCB mixtures are believed to result from activation of the same AhR-mediated pathways that provide the basis for the "dioxin-like" toxicity of certain PCB congeners. Because it is likely that the dioxin-like congeners represent a substantial portion of the potential toxicity of the total PCB mixture, application of USEPA's toxicological criteria for total PCBs to the remainder of the PCB mixture (i.e., after subtracting the dioxin-like congeners from the total), is not scientifically justifiable and will overstate risk for the remaining mixture. Because little is known about these non-dioxin-like congeners, the degree of overestimation cannot be determined.

In addition, the evaluation of PCBs using the TEQ approach requires that TEFs be used to convert measured concentrations of the 12 dioxin-like congeners to TEQ concentrations. There are substantial uncertainties associated with the TEFs that have been developed for these PCB congeners. These are due largely to several simplifying assumptions used in developing them, which are not well-supported in the scientific literature (Van den Berg et al. 2006; Roberts et al. 1990; Ema et al. 1994; Poland et al. 1994; Ramadoss and Perdew 2004; NAS 2006; Haws et al. 2006; Wiebel et al. 1996; Xu et al. 2000; Zeiger et al. 2001; Connor and Aylward 2006; Vamvakas et al. 1996; Silkworth et al. 2005; Carlson et al. 2009; Harper et al. 1995; Safe 1990; Starr et al. 1997; Toyoshiba et al. 2004; Walker et al. 2005; USEPA 2010e; SAB 2011). These include:

- The assumption that the dose-response curves for different congeners and endpoints are parallel.
- The assumption that the effects of multiple DLCs are additive.
- The assumption that humans are as sensitive as laboratory animals to the effects of DLCs.

- The assumption that noncancer endpoints and *in vitro* studies can be used to predict the carcinogenic potential of the individual DLCs.
- In addition, for a subset of PCB congeners, the TEF values were derived by comparing the toxicity of those congeners with that of 3,3′,4,4′,5-pentachlorobiphenyl (PCB-126) to develop relative effect potencies (REP) (Haws et al. 2006) rather than through direct comparison with TCDD. When developing REP estimates in this way, the principle of transitivity was invoked; that is, by quantifying both the toxicity of a DLC relative to PCB-126 and PCB-126 relative to TCDD, the toxicity of the DLC relative to TCDD can be estimated (USEPA 2010e). The TEF for PCB-126 was set at 0.1. Consequently, the PCB-126-based REPs were multiplied by 0.1 in the derivation of TEFs for other congeners in order to relate them to TCDD (Van den Berg et al. 2006). Given that the TEFs are meant to measure relative toxicity within an order of magnitude, and that two order-of-magnitude assumptions are being combined with this approach, this assumption could result in substantial over- or underestimation of actual toxicity of those PCB congeners. These issues are discussed in more detail in Appendix B.

Despite these issues, a secondary analysis was conducted to provide perspective on the estimated risks that would have resulted if the TEQ approach had been used instead to evaluate this subset of congeners. The concentrations of the dioxin-like PCB congeners were converted to TEQ_P concentrations, using the corresponding congener-specific TEFs, and the cancer risks from TEQ_P were evaluated using the cancer-based TDI for TCDD. The resulting risks were then added to the risks for TEQ_{DF} to derive a total risk for TEQ_{DFP}. In this approach, the carcinogenic potential of the remaining, non-dioxin-like PCBs was not calculated and added to the total.

When cancer hazards due to TEQ were calculated for the assumed consumption of biota by hypothetical recreational fishers,[30] estimated hazards were lower than the threshold of 1 for all scenarios. For the scenarios with the highest cancer hazard for biota consumption (e.g., those scenarios that assumed the consumption of catfish from FCA 2/3), the cancer hazard

[30] Comparisons of approaches could not be made for all pathways combined because PCB congeners were not analyzed in soils and sediments. As a result, the only media for which TEQ_{DFP} could be calculated and discussed were biota.

associated with TEQ_P was 0.13, the cancer hazard associated with TEQ_{DF} was 0.33, and the total cancer hazard for TEQ_{DFP} was 0.46 (Appendix H). The relative contributions of TEQ_{DF} and TEQ_P to total TEQ_{DFP} cancer hazard were 72 percent and 28 percent, respectively (Table 5-34).

It is more challenging to compare total PCB cancer risk with the TEQ cancer hazard because the two values are not comparable. However, if one uses the CSF approach to compare the relative cancer risks calculated for TEQ_{DF}, TEQ_P, and TEQ_{DFP} using USEPA's historical CSF for TCDD of 150,000 (mg/kg-day)$^{-1}$, a similar result is observed. As shown in Table 5-34, the total cancer risk using this approach was 3.6×10^{-5}, with TEQ_P contributing a risk of 9.9×10^{-6} and TEQ_{DF} contributing a risk of 2.6×10^{-5}. Thus, in this comparison, TEQ_{DF} also contributed 72 percent of the total risk. This is not surprising given that the relative concentrations of the individual congeners were the same, regardless of the toxicological criterion that was applied.

Results were somewhat different when the cancer risk for total PCBs (as the sum of 43 congeners), estimated using the USEPA CSF for PCBs, were compared with the estimated cancer risk for TEQ_{DF} using the same historical USEPA CSF. In this case, the cancer risk for total PCBs was 7.9×10^{-6}, the total cancer risk for TEQ_{DF} was 2.6×10^{-5} and the total combined cancer risk was 3.4×10^{-5}. Therefore, TEQ_{DF} contributed a slightly higher percentage of the total risk (77 percent).

This proportion changed considerably, however, depending on the CSF that was selected for evaluating the TEQ component. If the low end of the range of available CSFs (9,000 [mg/kg-day]$^{-1}$ based on FDA 1993) was used to evaluate TEQ_{DF}, then the relative risk contribution by total TEQ_{DF} was 16 percent. At the same time, if the upper end of the range of available CSFs (1,000,000 [mg/kg-day]$^{-1}$) was used to evaluate TEQ_{DF}, then TEQ_{DF} provided 96 percent of the total risk (Table 5-34). Therefore, if a linear dose response was used to evaluate TEQ_{DF}, the uncertainty about the correct CSF to be used to evaluate this mixture greatly complicates the interpretation of risk results.

IRIS does not discuss the approach to be used for evaluating noncancer effects of dioxin-like PCB congeners, so the same approach was used to evaluate the uncertainty associated with estimating noncancer effects of PCBs.

As shown in Table 5-34, when evaluating noncancer hazards, results varied depending upon the approach used. For recreational fish consumption under hypothetical Scenario 1A (i.e., direct contact with Beach Area A and consumption of catfish from FCA 2/3), the noncancer hazard for TEQ_{DF} was 1.1 and the noncancer hazard for TEQ_{P} was 0.42, for a total noncancer hazard for TEQ_{DF} of 1.52. Using this approach, TEQ_{DF} again contributed 72 percent of the total noncancer hazard. However, when the noncancer hazard for TEQ_{DF} (1.1) was combined with the noncancer hazard for Total PCBs (0.88), calculated using the RfD for Aroclor 1254, the total noncancer hazard was estimated to be 1.98 and TEQ_{DF} contributed only 56 percent of the hazard. This analysis indicated that the total PCB approach used to estimate noncancer hazards due to PCBs for this BHHRA resulted in higher (more conservative) estimates of the noncancer hazards associated with PCBs than would have been predicted if the TEQ_{DFP} approach had been used instead.

It should be noted, however, that there is no indication that the endpoints that were selected as the basis for the TCDD RfD are also associated with PCB toxicity. Thus, combining the dioxin-like PCBs with dioxins and furans to evaluate potential noncancer effects may be inappropriate, contributes uncertainty to the hazard estimates, and would make it likely that the endpoint-specific noncancer effects of TEQ_{DFP} would be overestimated.

5.2.5 Summary and Conclusions: Baseline Human Health Risk Assessment for the Area North of I-10 and Aquatic Environment

USEPA (1989) describes a human health risk assessment as a quantitative evaluation of the risk posed to human health by the actual or potential presence of chemicals in the environment. A risk assessment provides a conservative estimate of the likelihood of potential health effects in a specific hypothetical population that conforms to stated exposure assumptions, but it is a limited tool because it does not directly measure or predict the occurrence of any actual health effects in people who actually visit or use a site. The results of the risk assessment are intended to help site managers determine when remedial action is needed; determine health-protective levels of chemicals that may remain after remedial actions are completed; provide a basis for comparing the health impacts of remedial alternatives; and provide a consistent process for documenting risks (USEPA 1989).

For this BHHRA, risks were characterized for three hypothetical receptor groups: recreational fishers, subsistence fishers, and recreational visitors. The exposure media evaluated in the risk assessment were sediments in four individual beach areas, soils throughout the entire area of the northern impoundments and edible fish and shellfish that could be captured within USEPA's Preliminary Site Perimeter (i.e., hardhead catfish, clams, and crabs). For each receptor group, this BHHRA evaluated the potential for exposure to $COPC_{HS}$ in media within USEPA's Preliminary Site Perimeter, and the possibility that adverse health effects could occur as a result of assumed long-term exposures to these media under baseline conditions (i.e., immediately prior to the TCRA). The evaluation was completed for a series of different hypothetical scenarios that address direct contact in different areas or ingestion of different types of tissue from within USEPA's Preliminary Site Perimeter. In order to provide perspectives meaningful for comparing remedial alternatives, incremental risks from background, and reductions in risk resulting from completion of the TCRA, were also evaluated.

The parameters used for evaluating potential exposures and estimating risks and hazards relied on multiple conservative assumptions, which enhance the likelihood that potential assumed exposures and estimated risks are overestimated. The key findings of this BHHRA and conclusions about the potential health risks are summarized below.

Of the $COPC_{HS}$ identified for evaluation in this BHHRA for the area north of I-10 and the aquatic environment, dioxins and furans were identified as a risk driver in all media evaluated for the area north of I-10 and the aquatic environment. PCBs in fish and shellfish tissue, and methylmercury in catfish tissue were additionally identified as $COPC_{HS}$ that contributed substantially to potential risks associated with the area under study.

The results of this BHHRA generally indicate that hypothetical fishing and recreational exposure scenarios that assume direct contact with sediment within the original 1966 perimeter of the northern impoundments (i.e., termed "Beach Area E" throughout this risk assessment) under baseline conditions (i.e., immediately prior to the TCRA) would result in higher potential exposures to risk driving $COPC_{HS}$, than fishing and recreational scenarios elsewhere within the area under study.

To aid in the presentation of results in a manner useful for risk management, the results of the risk assessment are summarized in two sections below. First, the results for scenarios that assumed exposure to sediments at Beach Area E, together with consumption of fish or shellfish from the adjacent FCA, or soils from north of I-10 are summarized. Second, a summary of results for scenarios that assumed exposure to sediments at other areas (i.e., outside of the 1966 impoundment perimeter (termed Beach Area A, Beach Area B/C, and Beach Area D) in combination with consumption of fish or shellfish from adjacent FCAs or soils is presented.

5.2.5.1 Hypothetical Scenarios with Exposure at Beach Area E

Three types of hypothetical receptors—recreational fishers, subsistence fishers, and recreational visitors—with potential exposure to sediments at Beach Area E were evaluated. These scenarios assumed that recreational and subsistence fishers exposed via direct contact with beach sediments also ingested fish or shellfish from the adjacent FCA. Hypothetical recreational visitors who contacted sediments in this area were assumed to also contact soils throughout the study area.

5.2.5.1.1 Noncancer Hazards

RME noncancer HIs greater than 1 were estimated for hypothetical fishing and recreational scenarios that assume direct contact with sediments at Beach Area E. For all three potential receptor groups, regardless of the other media to which they were exposed, assumed direct contact to sediments in Beach Area E accounted for over 98 percent of the RME hazard for reproductive/developmental endpoints.[31] Although the HIs exceeded 1, these results do not necessarily indicate that adverse health effects would have occurred under baseline

[31] Reproductive/developmental endpoints were associated with exposure to dioxins and furans in all media, and methylmercury in catfish. For scenarios that included direct contact with sediments at Beach Area E, the HI for reproductive/developmental endpoints exceeded that for any other noncancer endpoint by more than an order of magnitude.

conditions. The CTE noncancer HIs for all potential receptors in this area were less than 1. The RME estimates relied on a number of highly conservative parameters, including the use of the maximum detected concentration of TEQ_{DF} as the EPC for estimating exposure. As a result, a substantial margin of safety was built into the RME estimates for the baseline condition. Completion of the TCRA construction in July, 2011 rendered sediments at Beach Area E inaccessible for direct contact by humans, and is also likely to have led to reductions in tissue concentrations in catfish and clams obtained from this area (although this cannot be confirmed with existing data), substantially reducing any baseline risks in this area.

5.2.5.1.2 Cancer Risks

All estimated excess cancer risks for potential recreational fishers, subsistence fishers, and recreational visitors who were assumed to contact $COPC_{HS}$ (other than dioxins and furans) in sediments and soils, and ingest fish or shellfish from the waters within USEPA's Preliminary Site Perimeter were within or below USEPA's target cancer risk range of 1×10^{-6} to 1×10^{-4}.

5.2.5.1.3 Cancer Hazards

RME dioxin cancer HIs greater than 1 were estimated for all hypothetical fisher and recreational visitor scenarios that assumed direct contact to sediments at Beach Area E. As was the case for noncancer hazards above, for these potential receptors assumed direct contact to sediment sediments in Beach Area E accounted for over 98 percent of the RME hazard. Although the cancer HIs exceeded 1, these results do not necessarily indicate that cancer effects to the hypothetical fishers and recreational visitors would have occurred under baseline conditions. The CTE cancer HIs for all hypothetical receptors in this area were less than 1, and the RME estimates relied on a number of highly conservative parameters, including the use of the maximum detected concentration of TEQ_{DF} as the concentration term for estimating exposure. As a result, a substantial margin of safety was built into the RME estimates. Completion of the TCRA construction in July, 2011 rendered sediments at Beach Area E inaccessible for direct contact by humans, substantially reducing any baseline risks in this area.

5.2.5.2 Scenarios with Exposure at Beach Areas A, B/C, and D

Three types of potential receptors with exposure to sediments at Beach Areas A, B/C, and D were evaluated. Hypothetical recreational and subsistence fishers exposed via direct contact with sediments at one of the defined beach areas were assumed to also ingest fish or shellfish from the adjacent FCA. Recreational visitors who contact sediments in one of the defined beach areas were assumed to also contact soils throughout the area under study.

5.2.5.2.1 Noncancer Hazards

This analysis indicated that no adverse noncancer health effects would be expected for hypothetical recreational visitors and recreational fishers as a result of contact with $COPC_{HS}$ in sediments at Beaches A, B/C, or D and soil throughout USEPA's Preliminary Site Perimeter, and consumption of fish or shellfish from the adjacent FCA. RME noncancer HIs for all $COPC_{HS}$ combined for hypothetical recreational fishers were below 1. For hypothetical recreational fishers, RME HIs grouped by toxicity endpoint, were all below 1.

Noncancer HIs greater than 1 occurred only for the hypothetical subsistence fisher under the following scenarios: direct contact to sediments at Beach Area A in combination with ingestion of catfish from the adjacent FCA 2/3; direct contact to sediments at Beach B/C in combination with consumption of either catfish from the adjacent FCA 2/3 or clams from the adjacent FCA 2; and direct contact to sediments at Beach D in combination with consumption of catfish from FCA 1.

For each of these scenarios the predominant pathway of estimated exposure was the consumption of tissue; direct contact with sediments accounted for less than 5 percent of exposure. Potential risk driving $COPC_{HS}$ in tissue were dioxins and furans and PCBs in catfish and clams, and methylmercury in catfish.

Although the noncancer HIs exceeded 1 in these scenarios, these results do not indicate that adverse health effects would have occurred in the hypothetical receptor group under baseline conditions. The RME estimates relied on a number of highly conservative parameters including upper bound consumption rates, the assumption that an individual would obtain 100 percent of the fish or shellfish consumed from the area under study over the entire assumed exposure duration, and the assumption that the concentration of lipophilic compounds would not be reduced through preparation or cooking.

As indicated by the PRA completed for this BHHRA, the influence of variability in estimated consumption rates and the portion of an individual's total consumption obtained from the area under study have large impacts on estimated exposures and resulting hazards for the hypothetical fisher population.

5.2.5.2.2 Cancer Risks

All estimated excess cancer risks for scenarios that assumed exposures to Beach Areas A, B/C, and D were within or below USEPA's target cancer risk range of 1×10^{-6} to 1×10^{-4}. These included both RME and CTE cancer risks for the hypothetical recreational fisher, subsistence fisher and recreational visitor scenarios.

5.2.5.2.3 Cancer Hazards

It is not expected that dioxin-related cancer effects would have occurred under the baseline hypothetical recreational visitor and recreational fisher scenarios as a result of assumed contact with dioxins and furans in sediments at Beach Area A, B/C, or D and soil, and consumption of fish or shellfish from within USEPA's Preliminary Site Perimeter. RME cancer TEQ_{DF} HIs for these potential receptor groups were all below 1.

RME dioxin cancer HIs greater than 1 were limited to the hypothetical subsistence fisher receptor group under the following assumed scenarios: direct contact with sediments at Beach Area A in combination with ingestion of catfish from the adjacent FCA 2/3; direct contact with sediments at Beach Area B/C in combination with consumption of catfish from the adjacent FCA 2/3; and direct contact with sediments at Beach D in combination with consumption of catfish from FCA 1.

For each of these hypothetical scenarios, consumption of tissue accounted for 95 percent or more of estimated $COPC_H$ exposure. Although the cancer HIs exceeded 1, these results do not indicate that cancer effects would have occurred in the hypothetical receptor group under baseline conditions. The RME estimates relied on a number of highly conservative parameters including upper-bound consumption rates, the assumption that an individual obtains 100 percent of the fish or shellfish consumed over the entire exposure duration from waters within USEPA's Preliminary Site Perimeter, and the assumption that concentrations of lipophilic compounds are not reduced during preparation or cooking.

5.2.5.3 Incremental Hazard

Exposure media that contributed the most to estimated human exposure to $COPC_{HS}$ included sediments at Beach Area E, catfish fillet at FCA 2/3 and FCA 1, and clams from FCA 2. However, risk-driving $COPC_{HS}$ present in catfish were also present at elevated concentrations in catfish harvested from background areas designated for this risk assessment. For example, in catfish fillet, 41 to 42 percent of the baseline hazard attributed to TEQ_{DF} exposures and 55 to 60 percent of baseline hazard associated with PCBs were also present under background conditions, suggesting that background conditions with respect to these $COPC_{HS}$ contributed roughly one-half of the total potential risks under relevant scenarios. In addition, the hazards associated with background exposure to methylmercury in catfish fillets were similar to or higher, indicating that any exposures from the study area are not contributing additional risks due to methylmercury.

5.2.5.4 Baseline Versus Post-TCRA Hazards

As discussed in detail in Appendix F, the post-TCRA noncancer TEQ_{DF} HIs for the hypothetical recreational fisher and recreational visitor scenarios are less than 1. For the hypothetical subsistence fisher, the exposure scenarios that assumed consumption of catfish in combination with direct contact to sediment (Scenarios 1A, 2A, and 3A) have post-TCRA RME TEQ_{DF} noncancer HIs of 6. These are lower than the baseline HIs, which ranged from 9 to 100, and higher than the background HIs of 4.

The post-TCRA cancer TEQ_{DF} HIs are less than 1 for all of the hypothetical recreational fisher and recreational visitor scenarios evaluated. Only the post-TCRA exposure scenarios for the hypothetical subsistence fisher that assumed consumption of catfish in combination with direct contact with sediment result in an RME cancer TEQ_{DF} HI of greater than 1 (HI=2). These are lower than baseline cancer TEQ_{DF} HIs, which ranged from 3 to 40, and only slightly higher than the background cancer TEQ_{DF} HIs of 1 for those scenarios.

The greatest hazard and risk reductions resulting from the TCRA are for baseline scenarios that assumed direct exposure to Beach Area E (Scenarios 3A, 3B, and 3C). This was because the majority of estimated TEQ_{DF} exposure and hazard for these scenarios was related to direct contact rather than to the ingestion of fish or shellfish, and because potential exposure to sediment in this area was completely restricted once the TCRA was implemented. For these scenarios, the hazard reductions resulting from TCRA implementation range from 84 to 100 percent. For hypothetical exposure scenarios that assumed direct contact with sediments at Beach Area A, B/C, or D and consumption of catfish or clam from the adjacent FCA, the hazard reductions resulting from the TCRA implementation range from 65 to 86 percent.

The post-TCRA evaluation indicated that the TCRA implementation has substantially reduced potential baseline risks for the area under study. Noncancer and cancer hazards calculated for the hypothetical recreational fisher and recreational visitor scenarios are all below the target HI of 1 under post-TCRA conditions. While potential noncancer and cancer hazards calculated for the hypothetical subsistence fisher scenario under post-TCRA conditions exceed the target HI of 1, these HIs exceed background levels only by factors of 2 or less.

6 EXPOSURE AND RISK CHARACTERIZATION FOR THE AREA OF INVESTIGATION ON THE PENINSULA SOUTH OF I-10

This section presents the exposure assessment and risk characterization for the area of investigation on the peninsula south of I-10. The purpose of the exposure assessment (Section 6.1) is to estimate the type and magnitude of potential human exposure to COPCs identified with respect to the area south of I-10 in the context of hypothetical exposure scenarios for a trespasser, commercial worker, and future construction worker. In the risk characterization (Section 6.2), these estimates of exposure are combined with toxicological criteria to yield numerical estimates of potential adverse health effects to a trespasser, a commercial worker, or a future construction worker exposed to the extent described by their respective exposure scenarios.

6.1 Exposure Assessment

For the area of investigation south of I-10, exposures were estimated using deterministic methods. The exposure scenarios, algorithms, and assumptions used for the deterministic assessment were established and discussed in the EAM (Appendix A) and are summarized below. This set of assumptions was used for calculating baseline exposures.

6.1.1 Exposure Scenarios

Two potential receptor groups were defined in the EAM for the quantitative risk assessment for the area of investigation on the peninsula south of I-10: a commercial worker, and a trespasser. USEPA comment 7 on the draft of this BHHRA requires that the BHHRA also evaluate risks that could result from exposures to soils greater than 2 feet deep because "construction-type activities may take place in this area in the future." Therefore, risks to a hypothetical future construction worker were also evaluated. Based on the CSM, updated to show this new hypothetical receptor for this area, the following hypothetical exposure scenarios were evaluated quantitatively:

- Trespasser—direct contact (incidental ingestion and dermal contact) with surface soil.
- Commercial Worker—direct contact (incidental ingestion and dermal contact) with surface and shallow subsurface soils.

- Future Construction Worker – direct contact (incidental ingestion and dermal contact) with surface and subsurface soils.

In estimating cumulative exposure for each potential receptor group, estimated exposures from the two direct contact pathways (i.e., ingestion and dermal contact) were summed.

6.1.1.1 Exposure Units

An exposure unit is defined in Section 5.1.1.1. To evaluate exposures of the hypothetical trespasser and hypothetical commercial worker to soils in the area of investigation south of I-10 (Figure 6-1), a single exposure unit was defined. This was based on the assumption that individuals trespassing or working in this area could have direct contact with soils in all of the sample collection areas during their visit. Because there is only a single exposure unit for these receptors in the area of investigation south of I-10, one hypothetical exposure scenario for the commercial worker scenario and one hypothetical exposure scenario for the trespasser were evaluated (Table 5-1).

Activities for hypothetical future construction workers may be confined to smaller areas than the entire area of investigation south of I-10. Therefore, for the evaluation of hypothetical future construction workers, smaller exposure units were developed. To identify the appropriate construction worker exposure units, construction worker-specific soil screening levels (SSLs) for $COPC_{HS}$ were calculated for soils 0 to 10 feet deep. These SSLs were derived using default exposure parameters for construction workers from USEPA (2002c) guidance, and chemical-specific inputs including noncancer and cancer toxicity criteria, RBA factors, and ABS_d outlined for this BHHRA in the TESM (Appendix B) and EAM (Appendix A). The exposure parameters were taken from USEPA's (2002c) *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites* and *Exposure Factors Handbook* (USEPA 2011a). USEPA's standard assumptions are conservative for the types and intensities of potential exposures that a hypothetical future construction worker could encounter, and are therefore appropriate for defining exposure units for this evaluation.

Construction worker SSLs were derived for cancer and noncancer endpoints and the lower (i.e., more conservative) of the two was adopted for identifying exposure units (Appendix M). The depth-weighted average of each $COPC_H$ at each soil core location was compared to the SSL. The depth-weighted average was used because a hypothetical future construction worker is assumed to be exposed to a mixture consisting of all soils within a 10-foot soil depth, and not solely to a given soil horizon for the duration of exposure. Any sample location with a depth-weighted average $COPC_H$ concentration exceeding the construction worker-specific SSL for one or more $COPC_{HS}$ was used to define the center of an individual exposure unit for the evaluation of soils from 0 to 10 feet deep.

To evaluate baseline risks to a hypothetical future construction worker, five 0.5-acre exposure units were identified using the screening process described above and detailed in Appendix M. USEPA (2002c) defines a default exposure unit of 0.5 acre for the evaluation of construction workers. In the absence of any specific information on how the area of investigation south of I-10 may be developed and the specific extent of construction work that may occur, this default exposure unit of 0.5 acre was adopted as the basis for the construction worker exposure assessment. Each soil core location with an exceedance was selected as an individual exposure unit because the sample density in these areas is approximately 0.5 acre. Figure 6-2 shows the five 0.5-acre exposure units selected for the evaluation of risk to a hypothetical future construction worker. Exposure units DS-1 (SJSB012), DS-2 (SJSB019), DS-3 (SJSB023), and DS-5 (SJSB025) were selected because the depth-weighted average concentration of TEQ_{DF} exceeded the construction worker SSL. Exposure unit DS-4 (SJSB022) was selected because the depth-weighted average concentration of arsenic exceeded the construction worker SSL.

The five selected exposure units for soils 0-10 feet deep in the area of investigation south of I-10 were used to evaluate five hypothetical construction worker exposure scenarios (Table 5-1).

6.1.2 Estimates of Exposure

This section presents the equations and exposure parameters that were used for estimating potential exposures for the area of investigation south of I-10. Both RME and CTE exposures were estimated.

6.1.2.1 Equations

Two types of exposures were evaluated: 1) ingestion of soil and 2) dermal contact with soil, as detailed below.

Equation 6-1. Intake via Ingestion of Soil

Relevant Receptor Groups: commercial worker, trespasser, and future construction worker

$$I_{soil} \frac{C_{soil} \times IR_{soil} \times RBA_{soil} \times FI_{soil} \times EF_{soil} \times ED \times CF_1}{BW \times AT} \qquad \text{(Eq. 6-1)}$$

Where:

I_{soil}	=	intake, the mass of a chemical contacted in soil by the receptor per unit body weight per unit time (mg/kg-day)
C_{soil}	=	chemical concentration in soil contacted over the exposure period (i.e., EPC for soil) (mg/kg)
IR_{soil}	=	soil ingestion rate (mg/day)
RBA_{soil}	=	relative bioavailability adjustment for soil (percent as a fraction)
FI_{soil}	=	fraction of total daily soil intake that is site-related (percent as a fraction)
EF_{soil}	=	exposure frequency (days/year)
ED	=	exposure duration (years)
CF_1	=	conversion factor (1×10^{-6} kg/mg)
BW	=	body weight (kg)
AT	=	averaging time (days)

Equations 6-2 and 6-3. Dermal Absorbed Dose via Contact with Soil

Relevant Receptor Groups: commercial worker, trespasser, and future construction worker

$$DAD_{soil} = \frac{DA_{event} \times SA \times EF_{soil} \times FI_{soil} \times ED \times EV}{BW \times AT} \quad \text{(Eq. 6-2)}$$

Where:

DAD_{soil}	=	dermal absorbed dose from soil (mg/kg-day)
DA_{event}	=	absorbed dose per event (mg/cm^2)
SA	=	skin surface area available for contact (cm^2)
EV	=	event frequency (day^{-1})

And

$$DA_{event} = (C_{soil} \times AF_{soil} \times F_{soil}) \times ABS_d \times CF_1 \quad \text{(Eq. 6-3)}$$

Where:

AF_{soil}	=	adherence factor for soil (mg/cm^2)
ABS_d	=	dermal absorption factor for soil (percent as a fraction)

6.1.2.2 Deterministic Exposure Evaluation

The EPCs and exposure parameters selected for each scenario are summarized below and are discussed in detail in the EAM (Appendix A).

6.1.2.2.1 Exposure Point Concentrations

For hypothetical trespassers and commercial worker scenarios (for which the entire area of investigation south of I-10 was defined as the single exposure unit for evaluation), EPCs were estimated for surface and subsurface soil according to the procedures outlined in Section 3.2. Table 6-1 summarizes the RME and CTE EPCs used for the deterministic assessment of baseline risks for the area of investigation south of I-10. Supporting documentation for the EPC derivations, including summaries of the best-fit distribution and basic summary statistics for each dataset, is provided as Appendix E.

For hypothetical future construction worker scenarios, for which five individual 0.5-acre units were defined as exposure units for evaluation, EPCs were estimated as the depth-weighted average concentration[32] of soils data at each individual sampling location. This depth-weighted average concentration was used for the RME and CTE exposure estimates, and reflects the fact that, in an actual exposure, the soils from 0 to 10 feet deep would be well mixed, and exposure to one small fraction of the soil for extended periods would not occur.

6.1.2.2.2 Exposure Parameters

This section provides an overview of the exposure assumptions used in the deterministic evaluation for the area of investigation south of I-10. A detailed presentation and the supporting rationales for these assumptions are included in the EAM (Appendix A) and a summary of these exposure parameters is presented in Table 6-2. Assumptions adopted for chemical-specific exposure parameters are provided in Table 5-7.

Common Parameters

For the hypothetical trespasser scenario, it was assumed that the trespasser is a young adult between the ages of 16 and 22 years. For the RME, the assumed exposure duration of 7 years was based on this assumed age group (16 to <23 years). For the CTE exposure, it was assumed that the trespasser visits the area of investigation on the peninsula south of I-10 for approximately one-half of the RME duration or 4 years. Because this area is currently fenced and actively managed for industrial activity, it is reasonable to assume that any activity would be infrequent. Therefore, an exposure frequency of 2 days per month or 24 days per year was assumed to evaluate the RME and 1 day per month or 12 days per year was assumed for the CTE. The mean body weight of 74 kg for males and females age 16 to <23 years was assumed for the trespasser (USEPA 2011a).

Commercial workers were assumed to be adults who perform work activities primarily outside. For the hypothetical commercial worker scenario, USEPA's (2002c) default exposure duration of 25 years was assumed for the RME and 12 years was assumed for the

[32] The method for calculating depth-weighted-averages is provided in the EAM (Appendix A) and in Appendix M to this BHHRA.

CTE. An exposure frequency of 225 days per year was assumed (USEPA 2002c). Based on USEPA (2011a), the mean body weight of 80 kg for male and female adults was used.

Hypothetical future construction workers were assumed to be adults that participate in soil excavation activities. For the hypothetical future construction worker scenario, USEPA's (2002c) default exposure duration of 1 year was assumed for the RME and CTE. An exposure frequency of 250 days per year and 125 days per year were assumed for the RME and CTE scenarios, respectively. The RME exposure duration was based on USEPA (2002c) guidance while the CTE was based on best professional judgment assuming an open excavation period of 6 months and a 5-day work week. A body weight of 80 kg was used (USEPA 2011a).

As discussed in Section 5.1.2.2.2, the averaging time depends on the toxic endpoint (cancer or noncancer) being assessed. For noncarcinogens, the averaging time was set equal to the exposure duration (e.g., for the hypothetical trespasser scenario with an assumed exposure duration of 7 years, the averaging time was 2,555 days). For carcinogens that were evaluated with a CSF, the averaging time was set equal to a lifetime (i.e., 78 years or 28,470 days) (USEPA 1989, 2011a). When the toxicity of a carcinogen was described using a criterion that assumes a threshold dose is required for an adverse effect to be elicited (i.e., TEQ_{DF}), then the averaging time was set equal to the exposure duration.

Parameters for Direct Contact

To evaluate incidental soil ingestion for the hypothetical trespasser scenario, an age-weighted soil ingestion rate of 41 mg/day was used for both the RME and CTE. This rate was based on USEPA's (2011a) recommended soil ingestion rate of 50 mg/day for individuals ages 6 to <21 years, and 20 mg/day for individuals age 21 and older. If, in fact, an individual does trespass in the area of investigation south of I-10, then it is anticipated that his or her stay would be for only a few hours at most. In addition, any such individuals likely would participate in daily activities at locations other than those locations in the area under study south of I-10 where exposure to soil could occur. In consideration of the likely short duration of daily activity in locations in the area of study compared to activities in other areas, fractional intakes for direct contact with soil of 0.5 and 0.25 were used for the RME and CTE, respectively.

To evaluate dermal contact for the hypothetical trespasser scenario, it was assumed that a trespasser's hands, forearms, lower legs, and feet might come into contact with surface soil. Based on this assumption and on the surface areas for these body parts provided in USEPA (2011a), a total surface area of 5,550 cm^2 was used to evaluate both the CTE and RME. Following USEPA recommendations, a body part-specific weighted adherence factor of 0.07 mg/cm^2 was calculated using data from a study of adults exposed to soil via a variety of soil activities. This adherence factor was used for both the CTE and RME.

For the hypothetical commercial worker scenario, it was assumed that the outdoor workers might be involved in contact-intensive activities. To account for the potentially more intensive contact, the recommended soil ingestion rate for outdoor workers of 100 mg/day was used for the RME (USEPA 2002c). Because workers might also be involved in less intensive activities, a rate of 50 mg/day was used to evaluate the CTE. This CTE rate is based on the recommended rate from USEPA (2002c) for an indoor worker. Because it is likely that some workers spend a majority of their time outdoors in the area of investigation south of I-10, the fractional daily intake of soil was assumed to be 1.0 for both RME and CTE.

To evaluate dermal contact for the hypothetical commercial worker scenario, it was assumed that a worker's head, forearms, and hands might come into contact with surface and shallow subsurface soil. Based on this assumption and surface areas for these body parts provided in USEPA (2011a), a total surface area of 3,470 cm^2 was used to evaluate both the CTE and RME. Following USEPA (2004) recommendation, a soil adherence factor of 0.2 mg/cm^2 was used and is based on data for a wide variety of activities during which an individual might be in contact with soil. This adherence factor was used for both the CTE and RME.

For the hypothetical future construction worker scenario USEPA's (2002c) default soil ingestion rate of 330 mg/day was used for the RME scenario. This value is based on the 95th percentile value for adult soil ingestion rate reported by Stanek et al. (1997), who completed a mass-balance tracer study in 10 individuals over 4 weeks duration. The variability in results obtained in the study for different trace elements and the short study duration make the estimate highly uncertain. These uncertainties are discussed further in the context of the results of the uncertainty evaluation below. For the CTE evaluation, a soil ingestion rate of 100 mg/day was adopted. This value is based on USEPA's default ingestion rate for outdoor workers, and is also used by the Massachusetts Department of Environmental Protection (1995) for evaluating exposure under construction scenarios. Because it is possible that construction workers could spend their entire work-day within the exposure unit of interest, the fractional intake of soil was assumed to be 1.0 for both RME and CTE estimates.

To evaluate dermal contact for the hypothetical future construction worker scenario, it was assumed that a worker's face, forearms, and hands might come into contact with surface and shallow subsurface soil. Based on this assumption and surface areas for these body parts provided in USEPA (2011a), a total surface area of 2,630 cm^2 was used to evaluate both the CTE and RME. Possible routine protective measures that could be taken by a future construction worker to protect their skin, such as gloves or other protective wear, are not considered by these assumptions. Following USEPA (2002c, 2004) recommendations, a soil adherence factor of 0.2 mg/cm^2 was used and is based on data for a wide variety of activities during which an individual might be in contact with soil. This adherence factor was used for both the CTE and RME.

Chemical-Specific Factors

In addition to the scenario-specific exposure assumptions described above, there are chemical-specific factors that were required to estimate $COPC_H$-specific exposure levels. Discussion of these chemical-specific factors was presented in Section 5.1.2.2.2 and summarized in Table 5-7. Further discussion of these parameters and the rationales for the values selected is presented in Appendix D.

6.2 Risk Characterization

As discussed in Section 5.2, risk characterization is the final step in the risk assessment process, where the goal is to present and interpret the key findings of the risk assessment, along with their limitations and uncertainties, for use in risk management decision-making. Three categories of health effects were evaluated for this BHHRA: cancer risk, noncancer hazard, and dioxin cancer hazard. Section 5.2.1 presents a general description of the methods used to estimate these potential effects. Very briefly, lifetime cancer risks in excess of background were calculated as the product the LADD and the CSF. Cancer risks in excess of background associated with each $COPC_H$ were summed across both of the assumed exposure routes (i.e., ingestion of soil and dermal contact with soil) and then across $COPC_{H}s$ to estimate overall excess cancer risk associated with potential exposures in the area of investigation on the peninsula south of I-10. Noncancer hazards (i.e., HQs) for each assumed exposure route were calculated as the ratio of the ADD to the RfD. Then the individual HQs for a given $COPC_{H}$ were summed for an individual receptor to derive a $COPC_{H}$-specific HI. Finally, the $COPC_{H}$-specific HIs were summed to derive a total HI for that exposure scenario. Consistent with USEPA guidance (1989) in the case that the total HI for a receptor exceeded 1 for all $COPC_{H}s$ combined, separate hazard indices for group of $COPC_{H}s$ that affect the same target organ or endpoint were estimated. These effect-specific HIs provide a more accurate indication of whether there is potential for a specific adverse health effect to occur to the potential receptors.

The carcinogenic potential for TEQ_{DF} was estimated using a hazard metric like that described for noncancer hazards above (Appendix B). Cancer hazards due to TEQ_{DF} were expressed as an HQ for a single assumed exposure route and an HI when hazards from all assumed exposure routes for a receptor were summed. Because cancer is a different toxic endpoint from the noncancer endpoints, the HIs for dioxin were not summed with noncancer hazards.

6.2.1 Baseline Risk Results for the Area of Investigation on the Peninsula South of I-10

This section presents the baseline deterministic risk results by potential receptor group for the area of investigation on the peninsula south of I-10. A summary of RME and CTE hazards and risks are provided in Table 6-3. The full set of risk and hazard estimates are provided as Appendix J, where Tables J-1 through J-3 present estimated exposures and resulting hazards and risks by exposure pathway, and Tables J-4 through J-6 present estimated hazards and risk by exposure scenario. Table J-7 shows the contribution of each $COPC_{H}$ and exposure pathway to overall risks and/or hazards for the hypothetical scenarios that resulted in excess cancer risk above $1{\times}10^{-4}$ or HIs greater than 1. These assumed relative contributions were used for identifying risk drivers.

6.2.1.1 Hypothetical Trespasser

The assumed exposure routes evaluated for the hypothetical trespasser are incidental ingestion and dermal contact with surface soil throughout the area of investigation south of I-10. Table 6-3 presents a summary of cumulative noncancer hazards, cancer risks, and TEQ_{DF} cancer hazards for the trespasser scenario. The noncancer RME HI is 0.006 and the CTE HI is 0.0004. The cumulative RME excess cancer risk is 2×10^{-7} and the CTE cancer risk is 9×10^{-9}. The RME TEQ_{DF} cancer HI for the hypothetical trespasser scenario is 0.0002, while the CTE TEQ_{DF} cancer HI is tenfold lower at 0.00002. Overall, for the hypothetical trespasser scenario, noncancer HIs and TEQ_{DF} cancer HIs are all less than 1. All estimated cancer risks in excess of background for this scenario were below USEPA's target cancer risk range of 1×10^{-6} to 1×10^{-4}.

6.2.1.2 Hypothetical Commercial Worker

Potential exposure routes for hypothetical commercial workers included incidental ingestion and dermal contact with surface and shallow subsurface soil. A single exposure scenario, which assumed direct exposure to soils throughout the area of investigation south of I-10, was evaluated for this potential receptor group.

Table 6-3 presents a summary of cumulative noncancer hazard, cancer risk, and dioxin cancer hazard for the hypothetical commercial worker scenario. The noncancer RME HI is 0.2, while the CTE HI is 0.04. The cumulative RME cancer risk is 3×10^{-5}. Cumulative CTE cancer risk is 3×10^{-6}. The RME TEQ_{DF} cancer HI is 0.006, while the estimated CTE TEQ_{DF} cancer HI is 0.002. Overall, for the hypothetical commercial worker scenario, noncancer HIs and TEQ_{DF} cancer HIs are all less than 1. All estimated excess cancer risks for this scenario are within USEPA's target cancer risk range of 1×10^{-6} to 1×10^{-4}.

6.2.1.3 Hypothetical Future Construction Worker

The assumed exposure routes evaluated for the hypothetical future construction worker are incidental ingestion and dermal contact with surface and subsurface soil. Five exposure units (DS-1 through DS-5) were evaluated.

Table 6-3 presents a summary of cumulative noncancer hazards, cancer risks, and TEQ_{DF} cancer hazards for the hypothetical future construction worker scenarios evaluated. The noncancer RME HIs ranged from 0.4 to 20 and noncancer CTE HIs ranged from 0.008 to 4. Table 6-4 presents endpoint-specific HIs for all hypothetical future construction worker scenarios that exhibited a HI greater than 1. All three scenarios with HI greater than 1 exhibited an endpoint-specific HI greater than 1. Scenarios DS-1, DS-2, and DS-4 had endpoint-specific RME HIs greater than 1. Scenarios DS-2 and DS-4 also had CTE HIs greater than 1. For these scenarios, TEQ_{DF} intake contributed over 99 percent of the estimated hazard (Appendix J).

Across all hypothetical future construction worker scenarios, cumulative estimated RME cancer risks (i.e., attributable to assumed exposure to arsenic, PCBs, and benzo(a)pyrene) ranged from 9×10^{-8} to 3×10^{-6}. Cumulative estimated CTE cancer risks ranged from 2×10^{-8} to 5×10^{-7} (Table 6-3).

RME TEQ_{DF} cancer HIs ranged from 0.004 to 7, while all CTE TEQ_{DF} cancer HIs were less than 1 (Table 6-3). Scenarios DS-2 and DS-4 exceeded a HI of 1.

Overall, hypothetical future construction worker scenarios that assumed direct contact at DS-1, DS-2, and DS-4 resulted in endpoint-specific (i.e., for reproductive/developmental effects) noncancer HIs greater than 1. Two of these same scenarios, specifically construction worker scenarios with assumed direct contact at DS-2 and DS-4, also resulted in TEQ_{DF} cancer hazard greater than 1. No cumulative cancer risks (i.e., attributable to assumed exposure to arsenic, PCBs, and benzo(a)pyrene) for these scenarios exceeded the 1×10^{-4} threshold (Table 6-3).

6.2.1.4 Summary of Deterministic Results

Hypothetical future construction worker scenarios evaluated for the area of investigation south of I-10 have endpoint-specific (i.e., reproductive/developmental endpoints) hazards and TEQ_{DF} cancer hazards greater than 1. No future construction worker scenarios evaluated have cancer risks greater than 1×10^{-4}. None of the hypothetical trespasser or commercial worker scenarios evaluated for the area of investigation south of I-10 have estimated cancer risks greater than 1×10^{-4}, endpoint-specific HIs greater than 1, or dioxin cancer HIs greater than 1. For scenarios with noncancer and TEQ_{DF} cancer hazards greater than 1, assumed TEQ_{DF} intake contributed over 99 percent of the estimated hazard. Therefore dioxins and furans are determined to be the sole risk-driving chemical for the area of investigation south of I-10.

6.2.2 Refined Analyses

Consistent with the approach summarized in Figure 1-4, additional analyses were considered to further characterize risks and/or hazards estimated for the hypothetical exposure scenarios that met one or more of the following thresholds:

- The cumulative estimated exposure from all pathways resulted in excess cancer risk $>1 \times 10^{-4}$
- The cumulative estimated exposure from all pathways resulted in a total endpoint-specific noncancer HI >1
- The cumulative estimated exposure from all pathways resulted in a dioxin cancer HI >1.

Although none of the scenarios included in the baseline deterministic evaluation for the area of investigation south of I-10 resulted in an estimated cancer risk greater than 1×10^{-4}, certain hypothetical scenarios for the future construction worker resulted in endpoint-specific HIs greater than 1 or dioxin cancer HIs greater than 1 (Table 6-3). The refined analyses considered for each selected scenario consisted of three evaluations: 1) an analysis and comparison of background hazards with the estimated deterministic hazards for the area under study, 2) an evaluation of post-TCRA hazards, and 3) a PRA of potential hazards.

However, no refined analyses were completed for the hypothetical future construction worker scenarios with resulting noncancer or cancer TEQ_{DF} HIs greater than 1. No background data for deeper soils were collected as part of the RI. In the absence of any background data for soils greater than 12 inches deep, a meaningful background comparison could not be completed. The TCRA implemented did not impact the area of investigation south of I-10. Therefore a post-TCRA evaluation was not relevant for this area. Finally, a PRA was not completed because the core samples providing the basis for the EPC in each hypothetical future construction worker exposure unit did not provide a sufficient number of results for a PRA.

6.2.3 Uncertainty Analysis

Risk characterization should present information important to interpreting risks in order to place the risk estimates in proper perspective. Uncertainties exist in each step of the risk assessment process, including the data collection and analysis, the estimation of potential exposures, and toxicity assessment. This section discusses the significant sources of uncertainty for the analysis.

6.2.3.1 Uncertainties in Data Treatment

Some uncertainty is introduced with the data rules applied in the calculation of EPCs. Following the data rules established for this assessment, TEQ_{DF} was calculated in two ways. First, individual congeners that were not detected in a sample were estimated to be present at one-half of the detection limit of that individual congener. Second, congeners that were not detected were treated as zero. The impact of the decision on the resulting TEQ_{DF} is dependent on both the number of congeners that were not detected and the detection limits for the congeners that were not detected. By comparing the resulting EPCs calculated using these two approaches, the impact of the uncertainty was determined. The difference in the EPCs for TEQ_{DF} applying one-half the detection limit to TEQ_{DF} applying zero were less than three percent (Table 6-1). Therefore, any uncertainty introduced by the treatment of non-detects does not substantially influence the risk results.

6.2.3.2 Uncertainties in Exposure Estimates

Minor exposure pathways that were not evaluated quantitatively include the inhalation of entrained dust derived from soil, and inhalation of volatile compounds present in soil. Generally, exposures to residents and commercial workers due to the inhalation of entrained dust originating from soils are considered to be orders of magnitude lower than exposures due to direct contact pathways (USEPA 2012c). Therefore, their contribution to overall risks associated with the hypothetical trespasser and hypothetical commercial worker scenarios is minimal. While inhalation of volatiles, if present, can contribute to total risk, none of the $COPC_{HS}$ identified is considered to be volatile.

Because there is a greater potential for dust to be generated during construction activities than in commercial settings, the potential contribution of inhalation exposures for hypothetical future construction workers was explored. For this analysis, the potential relative daily intake from inhalation of particulates to incidental ingestion of soil[33] was calculated, as:

$$Ratio(inhal/ingest) = \frac{BR}{IR_s \times PEF} \qquad \text{(Eq. 6-4)}$$

Where:

BR = breathing rate (m^3/day)

IRs = soil ingestion rate (kg/day)

PEF = particulate emission factor (m^3/kg).

This approach is presented by USEPA (2009c Appendix A) for determining the relative intakes via the ingestion and inhalation pathways for residents and commercial workers. It is used for this analysis with a future construction worker-scenario particulate emission factor (PEF), described further below.

The PEF represents an estimate of the relationship between chemical concentrations in soil and the concentration of these chemicals in air as a consequence of particulate suspension. Under a construction scenario, fugitive dusts may be generated from surface soils by wind erosion, construction vehicle traffic on temporary unpaved roads, and other construction activities. It is anticipated that the amount of fugitive dust is greater than the amount of dust generated under a residential or commercial worker scenario. USEPA guidance (2002c) provides an algorithm for estimating a PEF for construction worker scenarios. This algorithm models the dust generated from emissions from truck traffic on unpaved roads,

[33] The dermal pathway is not included in this comparison because: 1) based on default exposure assumptions, dermal exposure is relatively minor compared to oral exposure, and 2) it is difficult to include dermal exposure because it is expressed in terms of absorbed dose, while the oral and inhalation pathways are expressed in terms of administered dose.

which typically contribute the majority of dust emissions during construction. The algorithm is provided below:

$$PEF = \frac{Q}{C_{sr}} \times \frac{1}{F_d} \times \left[\frac{T \times Ag}{(Factor\ \ A \times ((365-p)/365) \times 281.9 \times \Sigma \mathrm{VKT}}\right] \qquad \text{(Eq. 6-5)}$$

Where:

Q/C_{sr}	=	Inverse of the ratio of the 1-hour geometric mean air concentration to the emission flux along a straight road segment bisecting a square site (g/m^2-s per kg/m^3)
F_d	=	Dispersion correction factor (unitless)
T	=	Total time over which construction occurs (seconds)
A_g	=	Surface area of contaminated road segment (m^2)
∑VKT	=	Sum of fleet vehicle kilometers traveled during the exposure duration (km)

$$Factor\ A = \frac{2.6 \times \left(\frac{s}{12}\right)^{0.8} \times \left(\frac{W}{3}\right)^{0.4}}{\left(\frac{M_{dry}}{0.2}\right)^{0.3}} \qquad \text{(Eq. 6-6)}$$

Where:

S	=	Road surface silt content (percent)
W	=	Mean vehicle weight (tons)
M_{dry}	=	Road surface material moisture content under dry, uncontrolled conditions (percent)

Using conservative default assumptions and regional information for Houston, a PEF of 1.7×10^7 m^3/kg was estimated for the construction worker scenario in the area of investigation south of I-10. Table 6-5 presents the assumptions used for the hypothetical future construction worker PEF.

Using Equation 6-5, the future construction worker PEF was used along with the RME soil ingestion rates and USEPA (2009c) defaults for worker inhalation rates to estimate the relative contribution of potential intakes from inhalation and incidental ingestion

(Table 6-6). The resulting ratio shows that the intake via inhalation is less than 1 percent of the intake via incidental ingestion. Therefore, significant uncertainty is not introduced into the risk assessment by treating inhalation exposures as a minor pathway that is not included in the quantitative calculation of exposure.

There are also some uncertainties associated with some of the assumptions used for estimating potential exposure via direct contact. For the area of investigation south of I-10, these include assumptions about exposure pattern and frequency for the hypothetical trespasser. The nature of trespassing is such that the activity is not expected to occur on a daily basis. The exposure frequency of 24 days or twice a month over the course of a year is a reasonable assumption. However, it is possible that trespassing activity could occur at a greater frequency. Even if a trespasser visited the area one day per week throughout the year, over the course of the exposure duration (i.e., 7 years for RME), risks and hazards would not exceed the risk thresholds set by USEPA of 1×10^{-4} and 1, respectively.

There are also uncertainties with some of the assumptions used for estimating potential exposures for the hypothetical future construction worker. Specific information about the construction activities that could occur in the future are not defined. These include the duration of any construction activities in which individuals may be exposed to soils as well as the specific location and extent of area that may be developed in the future. In the absence of this specific information for the area of study, conservative default assumptions from USEPA (2002c, 2011a) were adopted for estimating exposures for hypothetical future construction workers. If construction occurs in the future, it is possible that future construction workers may be exposed to concentrations of $COPC_{HS}$ in soils to a lesser extent than is assumed for this BHHRA. For example, if future construction workers are exposed for a lesser frequency than the 250 days per year assumed under the RME scenario or the 125 days per year assumed under the CTE scenario, their estimated exposures and estimated risks would also be reduced. If the area for construction were to be located at a different area than those designated by the exposure units DS-1 through DS-5 (Figure 6-2) or within a larger area, the potential exposures would also differ; the potential exposures, would likely be less if all other parameters were held constant. Because DS-1 through DS-5 were selected to represent assumed worst-case situations (i.e., to be based on the highest concentrations of

COCP$_{HS}$ in the area of investigation south of I-10), the potential exposures and risks estimated in this BHHRA also represent an upper bound to potential risks.

In addition, there is uncertainty in the upper-end adult soil ingestion rate reported by Stanek et al. (1997) and used for evaluating the RME hypothetical construction worker scenarios. Stanek et al. (1997) completed a mass-balance tracer study in 10 adults over a 4-week duration. The authors reported an average soil ingestion rate of 10 mg/day and a 95th percentile estimate of 331 mg/kg based on the best and most reliable four trace elements studied, but note also that, given the variability in the results of the trace elements used, there is substantial uncertainty in soil ingestion rates. The use of the incidental ingestion rate of 330 mg/day, based on this study, likely results in an overestimate of exposure to the hypothetical construction worker. Uncertainty exists, however, regarding the most appropriate estimated rate for incidental ingestion of soil by a hypothetical future construction worker.

6.2.3.3 Uncertainties in Toxicity Evaluation

The toxicity criterion that was used to evaluate potential cancer effects due to dioxins and furans (i.e., as TEQ$_{DF}$) was the TDI of 2.3 pg/kg-day derived from JECFA (2002). This TDI was developed based on the assumption that the cancer dose-response for TCDD and other DLCs is not linear and that there is a threshold for the carcinogenic effects of these compounds. There is substantial support for using a threshold approach to evaluate DLCs (WHO 1991, 1992, 1998; JECFA 2002; Simon et al. 2009; NAS 2006; ACC 2010; TCEQ 2010a,b, 2011; Haney 2010).

As discussed in Section 4.3.1, Section 5.2.4.3.1, and Appendix B, USEPA has been conducting its dioxin reassessment for nearly 20 years. While the scientific consensus during that period has been growing to conclude that DLCs act via a non-linear dose response, USEPA's most recent report on its reassessment indicates that it continues to assume that TCDD acts as a non-threshold carcinogen. Table 5-24 provides a summary of key toxicological criteria that have been developed by regulatory agencies and the scientific community for TCDD. These criteria are discussed in Section 5.2.4.3.1, and include criteria based on threshold and non-threshold (i.e., linear) models. Table 5-24 also presents RsD[34] derived using the CSFs. These RsDs can be compared to threshold based doses for cancer in order to provide perspective on the impact of different toxicity criteria on the risk results. Using the various CSFs results in RsDs ranging from 0.64 to 11 pg/kg-day when considering upper-bound Tier 3 CSFs ranging from 9,000 to 156,000 (mg/kg-day)$^{-1}$.

To meet requirements articulated by USEPA in comment 1 on the draft of this document (Appendix N), a sensitivity analysis of TEQ_{DF} cancer hazards and TEQ_{DF} cancer risks was completed. Tables 6-7 and 6-8 report RME and CTE TEQ_{DF} cancer hazards, respectively, for the area of study using the TDI of 2.3 pg/kg-day and TEQ_{DF} cancer risks with the CSF of 156,000 (pg/kg-day)$^{-1}$. To convey the cumulative impact of estimating TEQ_{DF} cancer risk with the CSF approach, cumulative cancer risks for other carcinogenic $COPC_{HS}$ and TEQ_{DF} are also shown. None of the construction worker scenarios evaluated resulted in TEQ_{DF} cancer risk or cumulative cancer risk from all $COPC_{HS}$ greater than $1{\times}10^{-4}$. Those scenarios with a TEQ_{DF} cancer HI greater than 1 (i.e., scenarios with assumed exposure to soils at DS-1, DS-2, and DS-4) had cumulative excess cancer risks ranging from $8{\times}10^{-6}$ to $3{\times}10^{-5}$.

Although USEPA has not established a CSF for assessment of dioxin cancer risk, there is substantial technical support for the use of the TDI instead of the CSF in risk assessment (Appendix B).

In addition, there are substantial uncertainties associated with USEPA's recently published RfD of 0.7 pg/kg-day for TCDD, which was used to evaluate the noncancer effects of DLCs in this BHHRA. This RfD was based on studies conducted by Baccarelli et al. (2008) and Mocarelli et al. (2008). Both evaluated health effects in human populations that were exposed to dioxins and furans as the result of a trichlorophenol reactor accident that occurred in 1976 in Seveso, Italy (USEPA 2012c). While this RfD has been adopted by USEPA, a number of questions arose during its peer review pertaining to the selection of appropriate NOAELs, pharmacokinetic consideration of increased elimination rates in children, correction for exposures to other dioxins and furans, and the full weight of evidence provided by other human and animal studies (SAB 2011; ACC 2010; Foster et al. 2010).

---

[34] The RsDs presented are based on a target risk level of $1{\times}10^{-4}$.

Differing values for noncancer effects have also been developed by other agencies worldwide. These are discussed above in Section 5.2.4.3.1 and Appendix B, and range from 1 to 4 pg/kg-day (DeRosa et al. 1999; Pohl et al. 2002; JECFA 2002). If any of these noncancer criteria were used to estimate noncancer effects in place of USEPA's recently published RfD of 0.7 pg/kg-day, the resulting noncancer hazards would be lower than those estimated and presented above (Table 6–3).

6.2.4 Summary and Conclusions: Baseline Human Health Risk Assessment for the Area of Investigation on the Peninsula South of I-10

For the area of investigation on the peninsula south of I-10, risks were characterized for three potential receptor groups: trespassers, commercial workers, and future construction workers. The exposure medium evaluated for this area was soil. For each scenario, potential exposures were evaluated via direct contact with soil (i.e., ingestion and dermal contact). For the hypothetical future construction worker, noncancer and TEQ_{DF} cancer HIs were greater than 1 for scenarios that assumed exposure to exposure units DS-1, DS-2, and DS-4. For these scenarios, over 99 percent of the estimated risk is attributable to assumed exposure to TEQ_{DF} in soils. For both the hypothetical commercial worker and trespasser scenarios, all cumulative risks are below 1×10^{-4} and noncancer and dioxin cancer hazards are below 1. The parameters used for evaluating potential exposures and estimating risks and hazards relied on multiple conservative assumptions, which enhance the likelihood that potential assumed exposures and estimated risks are overestimated.

7 REFERENCES

ACC, 2010. Technical comments on the derivation of cancer and noncancer toxicity criteria in EPA's reanalysis of key issues related to dioxin toxicity and response to NAS comments. Chlorine Chemistry Division of the American Chemistry Council. Comments to the EPA Science Advisory Board Dioxin Review Panel. July 9.

AECOM, 2012. Summary of cooking loss studies and data evaluation. Technical Memorandum submitted to U.S. Environmental Protection Agency, Region 2 on behalf of the Cooperating Parties' Group (CPG), Remedial Investigation/Feasibility Study, Lower Passaic River Study Area, Diamond Alkali Superfund Site, CERCLA Docket No. 02-2007-2009. July 5. 25 pp.

Alcoa, 1998. Draft Report for the Finfish/Shellfish Consumption Study, Alcoa (Point Comfort)/Lavaca Bay Superfund Site, Volume B7:Bay System Investigation Phase 2. Aluminum Company of America (ALCOA). January.

Anchor QEA and Integral, 2010. Final Remedial Investigation/Feasibility Study Work Plan, San Jacinto Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS, and Integral Consulting Inc., Seattle, WA. November 2010.

ATSDR, 2007. Toxicological Profile for Arsenic. U.S. Department of Health and Human Services, Public Health Service. Agency for Toxic Substances and Disease Registry. September. Available at http://www.atsdr.cdc.gov/toxpro2.html.

Baccarelli, A., S.M. Giacomini, C. Corbetta, M.T. Landi, M. Bonzini, D. Consonni, P Grillo, D.G. Pattterson, Jr., A.C. Pesatori, and P.A. Bertazzi, 2008. Neonatal thyroid function in Seveso 25 years after maternal exposure to dioxin. *PLoS Med* 5:e161. 44

Beauchamp, R., 2010. Personal communication (telephone conversation with P.N. Tomlinson, Integral Consulting Inc., Seattle, WA, on January 12, 2010, regarding observed recreational activities in vicinity of USEPA's Preliminary Site Perimeter). Texas Department of State Health Services, Austin, TX.

Bergstrom, C., J. Shirai, and J. Kissel, 2011. Particle Size Distributions, Size Concentration Relationships, and Adherence to Hands of Selected Geologic Media Derived From Mining, Smelting, and Quarrying Activities. *Sci. Total Environ*. 409(2011): 4247–4256.

Birnbaum, L.S., 1994. The Mechanism of Dioxin Toxicity: Relationship to Risk Assessment. *Environ Health Perspect* 102(Suppl 9):157-167.

Bukowski, J., L. Korn, and D. Wartenberg, 1995. Correlated Inuts in Quantitative Risk Assessment: The Effects of Distributional Shape. Risk Anal. 15(2):215–219.

Burmaster, D.E., 1997. LogNormal Distributions for Skin Area as a Function of Body Weight. Risk Analysis, 97-HE-015. June 14, 1997.

Burmaster, D.E., 1998. LogNormal distributions for Skin Area as a Function of Body Weight. Risk Analysis, 97-HE-015. Originally found in *Alceon*, 14 June 1997.

CalEPA, 1986. Technical Support Document. Report on Chlorinated Dioxins and Dibenzofurans. Part B - Health Effects of Chlorinated Dioxins and Dibenzofurans. California Environmental Protection Agency. Available online at: http://www.arb.ca.gov/toxics/id/summary/dioxptB.pdf

CalEPA, 2011. Air Toxics Hot Spots Program Risk Assessment Guidelines, Technical Support Document for Exposure Assessment and Stochastic Analysis, Public Review Draft. California Environmental Protection Agency, Office of Environmental Health Hazard Assessment, November.

Carlson, E.A., C. McCulloch, A. Koganti, S.B. Goodwin, T.R. Sutter, and J.B. Silkworth, 2009. Divergent transcriptomic responses to aryl hydrocarbon receptor agonists between rat and human primary hepatocytes. *Toxicological Sciences* 112(1):257-272.

Connor, K.T., and L.L. Aylward, 2006. Human response to dioxin: Aryl hydrocarbon receptor (AHR) molecular structure, function and dose-response data for enzyme induction indicate an impaired human AHR. *J. Toxicol Environ Health Part B* 9:147-171.

DeRosa, C.T., D. Brown, R. Dhara, W. Garrett, H. Hansen, J. Holler, D. Jones, D. Jordan-Izaguirre, R. O'Conner, H. Pohl, C. Xintaras, 1999. Dioxin and dioxin-like compounds in soil, Part I: ATSDR policy guideline and Part II: Technical support document for ATSDR policy guideline. *Toxicol Ind. Health* 15(6):552–576.

Duff, R.M. and J.C. Kissel, 1996. Effect of soil loading on dermal absorption efficiency from contaminated soils. *J. Toxicol. Environ. Health* 48:93–106.

Ebert, E.S., H.W. Harrington, K. Boyle, J. Knight, and R. Keenan, 1993. Estimating consumption of freshwater fish among Maine anglers. *North American Journal of Fisheries Management* 13:737-745.

Ema, M., N. Ohe, M. Suzuki, J. Mimura, K. Sogawa, S. Ikawa, and Y. Fujii-Kuriyama, 1994. Dioxin binding activities of polymorphic forms of mouse and human aryl hydrocarbon receptors. *J. Biol. Chem*. 269:27337–27343.

ENSR and EHA, 1995. Houston Ship Channel Toxicity Study. Prepared for the City of Houston, Houston, TX. ENSR Consulting and Engineering, Houston, TX and Espey, Huston and Associates, Austin, TX.

FDA, 1993. Report of the quantitative risk assessment committee. Subject: FAP OT4192, Update: Upper bound lifetime carcinogenic risks from exposure to dioxin congeners from foods contacting bleached paper products with dioxin levels not exceeding 2 ppt. U.S. Food and Drug Administration. January 27.

FDA, 1994. Polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans in bleached food-contact paper products; response to referral for action by the Environmental Protection Agency and Request for Comment. Federal Register 59(70):17384-7389. U.S. Food and Drug Administration. April 12.

FDA, 2013. Questions and Answers About Dioxins and Food Safety. Available online at: http://www.fda.gov/Food/FoodborneIllnessContaminants/ChemicalContaminants/DioxinsPCBs/ucm077524.htm. U.S. Food and Drug Administration.Foster, W.G., S. Maharaj-Briceno, and D.G. Cyr, 2010. Dioxin-induced changes in epididymal sperm count and spermatogenesis. *Environ Health Perspect.* 118:458-464.

Gentry, B., E. Wainwright, and D. Blankinship, 2005. Crystal Ball® 7.1 user manual. Decisioneering, Inc., Denver, CO. 357 pp.

Grieb, T., G. Bowie, C. Driscoll, S. Gloss, C Schofield, and D. Porcella, 1990. Factors Affecting Mercury Accumulation in Fish in the Upper Michigan Peninsula. *Environ. Toxicol. Chem*. 9(7):919-930.

Haney, J., 2010. Regulatory implications of USEPA's draft oral slope factor and reference dose for dioxin and the paradox of USEPA's surface soil draft interim preliminary remediation goal target risk/hazard levels for dioxin versus dioxin risk/hazard from typical dietary exposure and breast milk intake. Texas Commission on Environmental Quality (TCEQ) Comments to the Science Advisory Board (SAB) Dioxin Review Panel for the October 27-29, 2010 Public Meeting, Washington, DC.

Harper, N., K. Connor, M. Steinberg, and S. Safe, 1995. Immunosuppressive activity of polychlorinated biphenyl mixtures and congeners: nonadditive (antagonistic) interactions. *Fundamental and Applied Toxicology* 27:131-139.

Haws, L.C., S.H. Su, M. Harris, J. DeVito, N.J. Walker, W.H. Farland, B. Finley, and L.S. Birnbaum, 2006. Development of a refined database of mammalian relative potency estimates for dioxin-like compounds. Toxicol. Sci. 89(1):4-30.

Helsel, D.R., 2005. *Nondetects and data analysis: statistics for censored environmental data*. John Wiley & Sons, Inc., Hoboken, NJ. 268 pp.

Integral, 2010a. Sampling and Analysis Plan (SAP): Soil Study, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA.

Integral, 2010b. Sampling and Analysis Plan: Tissue Study, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA.

Integral, 2010c. Technical Memorandum on Bioaccumulation Modeling, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. September 2010.

Integral, 2011a. COPC Technical Memorandum, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. May 2011.

Integral, 2011b. Draft Addendum 3 to the Soil Sampling and Analysis Plan (SAP) for additional Soil Sampling South of Interstate Highway 10 (I-10), San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. December, 2011.

Integral, 2011c. Soil Sampling and Analysis Plan, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS, and Integral Consulting Inc., Seattle, WA. January 2011.

Integral, 2012a. Exposure Assessment Memorandum, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. May.

Integral, 2012b. Toxicological and Epidemiological Studies Memorandum, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. May.

Integral and Anchor QEA, 2010. Sampling and Analysis Plan: Sediment Study San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS, and Integral Consulting Inc., Seattle, WA. April 2010.

Integral and Anchor QEA, 2012a. Draft Remedial Investigation Report, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS, and Integral Consulting Inc., Seattle, WA. November 2012.

Integral and Anchor QEA, 2012b. Preliminary Site Characterization Report, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS, and Integral Consulting Inc., Seattle, WA. February, 2012.

ITRC. 2011. Incorporating Bioavailability Considerations into the Evaluation of Contaminated Sediments. Interstate and Technology Regulatory Council.

JECFA, 2002. Polychlorinated dibenzodioxins, polychlorinated dibenzofurans, and coplanar polychlorinated biphenyls. WHO Food Additives Series 48. Available online at: http://www.inchem.org/documents/jecfa/jecmono/v48je20.htm. Joint FAO/WHO Expert Committee on Food Additives.

Johnson, G., R. Ehrlich, W. Full, and S. Ramos, 2007. Principal component analysis and receptor models in environmental forensics. pp. 207–261. In: *Introduction to environmental forensics.* Second Edition. Brian Murphy, Robert D. Morrison (eds.). Academic Press.

Keenan, R.E., D.J. Paustenbach, R.J. Wenning, and A.H. Parsons, 1991. A pathology re-evaluation of the Kociba et al. (1978) bioassay of 2,3,7,8-TCDD: implications for risk assessment. *J. Toxicol. Environ. Health* 34:279-296.

Kociba, R.J., D.G. Keyes, J.E. Beyer, R.M. Carreon, C.E. Wade, D.A. Dittenber, R.P. Kalnins, L.E. Frauson, C.N. Park, S.D. Barnard, R.A. Hummel, and C.G. Humiston, 1978. Results of a two-year chronic toxicity and oncogenicity study of 2,3,7,8-tetrachlorodibenzo-p-dioxin in rats. *Toxicol. Appl. Pharmacol.* 46:279-303.

Koenig, L., 2010. Personal Communication (telephone conversation with D. Rudnick, Integral Consulting Inc., Seattle, WA, on March 12, 2010, regarding sediment PCB data for San Jacinto). Texas Commission on Environmental Quality.

Lange, T., H. Royals, and L. Connor, 1993. Influence of water chemistry on mercury concentration in largemouth bass from Florida lakes. *Trans. Am. Fish. Soc.* 122(1):74-84.

MacFarland, V.A. and J.U. Clarke, 1989. Environmental occurrence, abundance, and potential toxicity of polychlorinated biphenyl congeners: considerations for a congener-specific analysis. *Environmental Health Perspectives* 81:225-239.

Massachusetts Department of Environmental Protection, 1995. Calculation of an enhanced soil ingestion rate. Technical update. Massachusetts Department of Environmental Protection, Boston, MA. 3 pp.

Mocarelli, P., P.M. Gerthoux, D.G. Patterson, Jr., S. Milani, G. Limonta, M. Bertona, S. Signorini, P. Tramacere, L. Colombo, C, Crespi, P. Brambilla, C. Sarto, V. Carreri, E. Sampson, W.E. Turner, and L. Needham, 2008. Dioxin exposure, from infancy through puberty, produces endocrine disruption and affects human semen quality. *Environmental Health Perspectives* 116:70-77.

NAS, 2006. Health Risks from Dioxin and Related Compounds: Evaluation of the EPA Reassessment. National Academy of Sciences, Committee on EPA's Exposure and Human Health Reassessment of TCDD and Related Compounds, National Research Council. Washington, DC.

Newman, J.W., J.S. Becker, G. Glondina, and R.S. Tjeerdema, 1998. Quantitation of Aroclors Using Congener-Specific Results. *Envrion. Tox. Chem*. 17(11): 2159–2167.

NOAA, 2008. Mean number of days with precipitation .01 inch or more. National Oceanic and Atmospheric Administration.

NRC, 2006. Health Risks from Dioxin and Related Compounds: Evaluation of the EPA Reassessment. National Research Council. Available online at: ttp://www.nap.edu/catalog.php?record_id=11688. Accessed January 23, 2008.

NTP, 1982. Carcinogenesis Bioassay of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (CAS No. 1746-01-6) in Osborne-Mendel rats and B6C3F1 mice (Gavage Study). Technical Report Series, Issue 209:195. National Toxicology Program.

NTP, 2006. NTP Technical Report on the Toxicology and Carcinogenesis Studies of 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin (TCDD) in Female Harlan Sprague-Dawley rats. NTP TR 521. National Toxicology Program.

OEHHA, 2007. Adoption of the Revised Air Toxics Hot Spots Program Technical Support Document for Cancer Potency Factors (06/01/09). California Office of Environmental Health and Hazard Assessment (OEHHA). http://www.oehha.ca.gov/air/hot_spots/tsd052909.html

Pandelova, M., B. Henkelmann, O. Roots, M. Simm, L. Järv, E. Benefenati, and K.-W. Schramm, 2008. Levels of PCDD/F and Dioxin-like PCB in Baltic Fish of Different Age and Gender. *Chemosphere* 71:369-378.

Pohl, H.R., H.E. Hicks, D.E. Jones, H. Hansen, and C.T. De Rosa, 2002. Public health perspectives on dioxin risks: Two decades of evaluations. *HERA*. 8:233–250.

Poland, A., D. Palen, and E. Glover, 1994. Analysis of the four alleles of the murine aryl hydrocarbon receptor. Mol. Pharmacol. 46(5):915-921.

Portier, K., J.K. Tolson, and S.M. Roberts, 2007. Body Weight Distributions for Risk Assessment. *Risk Analysis* 27(1):11-26.

Ramadoss, P., and G.H. Perdew, 2004. Use of 2-azido-3-[125I]iodo-7,8-dibromodibenzo-pdioxin as a probe to determine the relative ligand affinity of human versus mouse aryl hydrocarbon receptor in cultured cells. *Mol. Pharmacol*. 66(1):129-136.

Roberts, E.A., K.C. Johnson, P.A. Harper, and A.B. Okey, 1990. Characterization of the Ah receptor mediating aryl hydrocarbon hydroxylase induction in the human liver cell line Hep G2. *Arch. Biochem. Biophys*. 276(2):442-450.

Roberts, S.M., J.W. Munson, Y.W. Lowney, and M.V. Ruby, 2007. Relative oral bioavailability of arsenic from contaminated soils measured in the cynomolgus monkey. *Toxicol. Sciences* 95(1): 281-288.

Roots, O., and V. Zitko, 2006. The Effect of Age on the Concentration of Polychlorinated Dibenzo-p-Dioxins, Dibenzofurans and Bioxin-like Polychlorinated Biphenyls in Baltic Herring and Sprat. *Fresenius Environ. Bull*. 15(3):207-219.

Roy, T.A., K. Hammerstrom, and J. Schaum, 2008. Percutaneous Absorption of 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin (TCDD) from soil. *J. of Tox. Env. Health*. Part A. 71 (23):1509-1515.

SAB, 2011. SAB Review of EPA's Reanalysis of Key Issues Related to Dioxin Toxicity and Response to NAS Comments (May 2010). U.S. Environmental Protection Agency, Science Advisory Board. SAB-011-014. August.

Safe, S., 1990. Polychlorinated biphenyls (PCBs), dibenzo-*p*-dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: Environmental and mechanistic considerations which support the development of toxic equivalency factors (TEFs). *Crit. Rev. Toxicol.* 21(1):51-88.

Shoaf, M.B., J.H. Shirai, G. Kedan, J. Schaum, J.C. Kissel, 2005. Child dermal sediment loads following play in a tide flat. *Journal of Exposure Analysis and Environmental Epidemiology* 15(5):407-412.

Shu, H., P. Teitelbaum, A.S. Webb, L. Marple, B. Brunck, D. Dei Rossi, F.J. Murray, and D. Paustenbach, 1988. Bioavailability of Soil-Bound TCDD: Dermal Bioavailability in the Rat. *Fund. App. Toxicol.* 10:335–343.

Silkworth, J.B., A. Koganti, K. Illouz, A. Possolo, M. Zhao, and S.B. Hamilton, 2005. Comparison of TCDD and PCB CYP1 induction sensitivities in fresh hepatocytes from human donors, Sprague-Dawley rats, and rhesus monkeys and HepG2 cells. *Toxicol. Sci.* 87(2):508-519.

Simon, T., L.L. Aylward, C.R. Kirman, J.C. Rowlands, and R.A. Budinsky, 2009. Estimates of cancer potency of 2,3,7,8-tetrachlorodibenzo(p)dioxin using linear and nonlinear dose-response modeling and toxicokinetics. *Toxicological Sciences* 112(2):490-506.

Singh, A., R. Maichle, and S.E. Lee, 2006. On the Computation of a 95% Upper Confidence Limit of the Unknown Population Mean Based Upon Data Sets with Below Detection Limit Observations. EPA/600/R-06/022. Lockheed Martin Environmental Services, Contract No. 68-W-04-005. Task Order No. 09. Las Vegas, NV.

Smith, A.E., P.B. Ryan, J.S. Evans, 1992. The effect of neglecting correlations when propating uncertainty and estimating the population distribution of risk. *Risk Anal.* 12(4): 467–474.

Spalt, E.W., J.C. Kissel, J.H. Shirai, and A.L. Bunge, 2009. Dermal absorption of environmental contaminants from soil and sediment: a critical review. *J. Exp. Sci. Environ. Epid.* 19:119–148.

Stanek, E.J., E.J. Calabrese, R. Barnes, and P. Pekow, 1997. Soil ingestion in adults – results of a second pilot study. *Ecotoxicol. Environ. Saf.* 36: 249-257

Stanek, E.J., E.J. Calabrese, and R. Barns, 1999. Soil ingestion estimates for children in Anaconda using trace element concentrations in different particle size fractions. *Hum. Ecol. Risk. Assess* 5(3):547-558.

Stanek, E.J., E.J. Calabrese, and M. Zorn, 2001. Biasing factors for simple soil ingestion estimates in mass balance studies of soil ingestion. *Human and Ecological Risk Assessment.* 7(2):329-325.

Starr, T.B., T.R. Zacharewski, T.R. Sutter, S.H. Safe, W.F. Greenlee, and R.B. Connolly, 1997. Concerns with the use of a toxicity equivalence factor (TEF) approach for risk assessment of "dioxin-like" compounds. *Organohalogen Compounds* 34:91-94.

TCEQ, 2009. Toxicity factors and chemical/physical parameters. TCEQ Regulatory Guidance. Texas Commission on Environmental Quality, Remediation Division. RG-366/TRRP-19. March.

TCEQ, 2010a. Comments Regarding the U.S. Environmental Protection Agency "Draft EPA's Reanalysis of Key Issues Related to Dioxin Toxicity and Response to NAS Comments." Notice of Public Comment Period 75 FR 28610, May 21, 2010. Docket ID No. EPA-HQ-IRD-2010-0395.

TCEQ, 2010b. Texas Commission on Environmental Quality Comments Regarding the U.S. Environmental Protection Agency Draft Recommended Interim Preliminary Remediation Goals for Dioxin in Soil at CERCLA and RCRA Sites, Notice of Availability and Announcement of Public Comment Period, 75 FR 0984, January 7, 2010, Docket ID No. EPA-HQ-SFUND-2009-0907. Submitted on February 26, 2010.

TCEQ, 2011. Guidelines to Develop Inhalation and Oral Cancer and Non-cancer Toxicity Factors. Peer Review Draft. June 7. Available at: www.tera.org/peer/tceqesl. Texas Commission on Environmental Quality, Austin, TX.

TCEQ and USEPA, 2006. Screening Site Assessment Report San Jacinto River Waste Pits, Channelview, Harris County, Texas. TXN000606611. Texas Commission on Environmental Quality and U.S. Environmental Protection Agency.

TDSHS, 2007. Standard Operating Procedures and Quality Control/Assurance Manual. Texas Department of State Health Services, Seafood and Aquatic Life Group, Austin, TX. February 2007.

TDSHS, 2008. Characterization of Potential Adverse Health Effects Associated with Consuming Fish or Blue Crab from Trinity Bay and Upper Galveston Bay. Chambers, Galveston, and Harris Counties, Texas. Texas Department of State Health Services, Seafood and Aquatic Life Group, Policy, Standards, and Quality Assurance Unit and Regulatory Services Division. April 2008

TDSHS, 2010. Texas Fish Tissue Data. Collection of Excel files sent to Jennifer Sampson (Integral) from Michael Tennant (TDSHS) on 1/20/2010 containing tables of fish tissue chemical data collected over several decades from the Galveston Bay area. Texas Department of State Health Services.

TDSHS, 2012. Public Health Assessment, Final Release. San Jacinto River Waste Pits, Channelview, Harris County, Texas. Prepared for U.S. Department of Health and Human Services, Agency for Toxic Substances and Disease Registry, Division of Health Assessment and Consultation, Atlanta, GA.

Toyoshiba, H., N.J. Walker, A.J. Bailer, and C.J. Portier, 2004. Evaluation of toxic equivalency factors for induction of cytochromes P450 CYP1A1 and CYP1A2 enzyme activity by dioxin-like compounds. *Toxicol. Appl. Pharmacol*. 194:156–168.

University of Houston and Parsons, 2006. Total Maximum Daily Loads for Dioxins in the Houston Ship Channel. Contract No. 582-6-70860, Work Order No. 582-6-70860-02. Quarterly report No. 3. Prepared in cooperation with the Texas Commission on Environmental Quality and the U.S. Environmental Protection Agency. University of Houston and Parsons Water & Infrastructure. Available online at: http://www.tceq.state.tx.us/assets/public/implementation/water/tmdl/26hscdioxin/26-all-data-compiled-q3-fy06.pdf.

University of Houston and Parsons, 2009. Total Maximum Daily Loads for PCBs in the Houston Ship Channel. Contract No. 582-6-60860. Work Order No. 528-6-60860-19. Draft Final Report. Prepared in cooperation with the Texas Commission on Environmental Quality and the U.S. Environmental Protection Agency. University of Houston and Parsons Water & Infrastructure.

URS, 2010. Data Usability Summary. Surface Water and Sediment Samples. San Jacinto River Waste Pits Superfund Site, Channelview, Harris County, Texas. Prepared for Texas Commission on Environmental Quality, Austin, Texas. Project No: 25335373. URS Corporation, Houston, TX.

USCB, 2012. American Fact Finder. Available online at: http://factfinder2.census.gov/faces/nav/jsf/pages/index.xhtml U.S. Census Bureau, Washington, DC.

USEPA, 1985. Health Assessment Document for Polychlorinated Dibenzo-*p*-Dioxins. U.S. Environmental Protection Agency.

USEPA, 1988. Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

USEPA, 1989. Risk Assessment Guidance for Superfund (RAGS): Volume 1 – Human Health Evaluation Manual (Part A), Interim Final. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

USEPA, 1990. National contingency plan. 55 Fed. Reg. 8665-8865 (Mar. 8, 1990). U.S. Environmental Protection Agency, Washington, DC.

USEPA, 1991a. Risk Assessment Guidance for Superfund (RAGS): Volume 1 – Human Health Evaluation Manual (Part B, Development of Risk-Based Preliminary Remediation Goals), Interim. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC. EPA/540/R-92/003.

USEPA, 1991b. Risk Assessment Guidance for Superfund: Volume 1 – Human Health Evaluation Manual (Part C, Risk Evaluation of Remedial Alternatives). Interim. Publication 9285.7-01C. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC. October.

USEPA, 1991c. Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual, Supplemental Guidance, "Standard Default Exposure Factors" Interim Final. PB91-921314. OSWER Directive: 9285.6-03. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Toxics Integration Branch, Washington, DC.

USEPA, 1993. Superfund's Standard Default Exposure Factors for the Central Tendency and Reasonable Maximum Exposure. EPA/600-D-93-901. U.S. Environmental Protection Agency. November 4.

USEPA, 1996. Soil Screening Guidance: User's Guide. Publ. No. 9355.4-23. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC. July. USEPA 2011A. Exposure Factors Handbook 2011 Edition. EPA/600-R-09/052F. U.S. Environmental Protection Agency, National Center for Environmental Assessment, Office of Research and Development, Washington, DC. September.

USEPA, 1997a. Exposure Factors Handbook - Volume 1. General Factors. U.S. Environmental Protection Agency, National Center for Environmental Assessment, Office of Research and Development, Washington, DC; Versar Inc., Exposure Assessment Division (Springfield, VA). EPA/600/P-95/002Fa. August.

USEPA, 1997b. Health Effects Assessment Summary Tables. FY 1997 Update. 9200.6-303 (97-1). EPA-540-R-97-036. PB97-921199. U.S. Environmental Protection Agency, Office of Research and Development, Office of Emergency and Remedial Response, Washington, DC. July.

USEPA, 1998. Guidelines for ecological risk assessment. EPA/630/R095/002F. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC. 188 pp

USEPA, 2000a. Draft Exposure and Human Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related compounds. National Center for Environmental Assessment, U.S. Environmental Protection Agency, Washington, DC. Accessed at http://cfpub1.epa.gov/ncea/cfm/part1and2.cfm?ActType=default.

USEPA, 2000b. Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories. Volume 2. Risk Assessment and Fish Consumption Limits. Third edition. EPA 823-B-00-008. Appendix C-1. U.S. Environmental Protection Agency, Office of Water. Washington, DC.

USEPA, 2001. Risk Assessment Guidance for Superfund (RAGS): Volume III—Part A: Process for Conducting Probabilistic Risk Assessment. EPA-540-R-02-002. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

USEPA, 2002a. Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites. 9285.6-10. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2002b. Estimated Per Capita Fish Consumption in the United States. EPA/821/C-02/003. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

USEPA, 2002c. Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites. OSWER 9355.4-24. U.S. Environmental Protection Agency, Solid Waste and Emergency Response, Washington, DC.

USEPA, 2003a. Human Health Toxicity Values in Superfund Risk Assessments. OSWER Directive 9285.7-53. Available online at: http://www.epa.gov/oswer/riskassessment/pdf/hhmemo.pdf. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2003b. Technical Summary of Information Available on the Bioaccumulation of Arsenic in Aquatic Organisms. EPA822R03032. Available online at: http://www.epa.gov/waterscience/criteria/arsenic/tech-sum-bioacc.pdf. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2004. Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment). U.S. Environmental Protection Agency, Office of Superfund Remedial and Technology Innovation, Washington, DC.

USEPA, 2005. Guidelines for Carcinogen Risk Assessment. EPA/630/P-03/001F. Risk assessment forum, U.S. Environmental Protection Agency.

USEPA, 2008. Framework for Application of the Toxicity Equivalence Methodology for Polychlorinated Dioxins, Furans, and Biphenyls in Ecological Risk Assessment. EPA 10/R-0 /00-200.USEPA, 2008b. Child-Specific Exposure Factors Handbook. EPA/600/R-06/096F. National Center for Environmental Assessment, Office of Research and Development, Washington, DC, September.

USEPA, 2009a. The National Study of Chemical Residues in Lake Fish Tissue. EPA-823-R-09-006. U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology. September.

USEPA, 2009b. Public Review Draft. Draft Recommended Interim Preliminary Remediation Goals for Dioxin in Soil at CERCLA and RCRA Sites. OSWER 9200.3-56. U.S. Environmental Protection Agency, Office of Superfund Remediation and Technology Innovation, Washington, DC. December 30.

USEPA, 2009c. Unilateral Administrative Order for Remedial Investigation/Feasibility Study. U.S. EPA Region 6 CERCLA Docket No. 06-03-10. In the matter of: San Jacinto River Waste Pits Superfund Site Pasadena, Texas. International Paper Company, Inc. & McGinnes Industrial Management Corporation, respondents.

USEPA, 2010a. Decision Document for the Time Critical Removal Action at the San Jacinto River Waste Pits Site, Harris County, Texas. USEPA Region 6. July 28, 2010.

USEPA, 2010b. Draft EPA's Reanalysis of Key Issues Related to Dioxin Toxicity and Response to NAS Comments. EPA/600/R-10-038A. Available at www.epa.gov/iris. U.S. Environmental Protection Agency, Office of Research and Development, Cincinnati, OH.

USEPA, 2010c. Guidance for Implementing the January 2001 Methylmercury Water Quality Criterion. Final. EPA-823-R-10-001. U.S. Environmental Protection Agency, Office of Science and Technology, Washington, DC. April 2010.

USEPA, 2010d. Final Report, Bioavailability of Dioxins and Dioxin-Like Compounds in Soil; prepared for U.S. Environmental Protection Agency, Office of Superfund Remediation and Technology Innovation, Environmental Response Team – West, Las Vegas, NV. Prepared by SRC, Inc., Chemical, Biological and Environmental Center, N. Syracuse, NY.

USEPA, 2010e. Recommended Toxicity Equivalence Factors (TEFs) for Human Health Risk Assessments of 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxins and Dioxin-Like Compounds. EPA/100/R-10/005. U.S. Environmental Protection Agency, Risk Assessment Forum. Washington, DC.

USEPA, 2010f. Relative Bioavailability of Arsenic in Soils at 11 Hazardous Waste Sites Using an *In Vivo* Juvenile Swine Method. OSWER Directive #9200.0-76. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Bioavailability Subcommittee of the Technical Review Workgroup, Washington, DC. June 2010.

USEPA, 2011a. Exposure Factors Handbook 2011 Edition. EPA/600-R-09/052F. U.S. Environmental Protection Agency, National Center for Environmental Assessment, Office of Research and Development, Washington, DC. September.

USEPA, 2011b. EPA Splits Dioxin Risk Estimate after Divided Review from Science Advisors. Inside EPA, U.S. Environmental Protection Agency. September 2. Available at INSIDEEPA.com Baccarelli et al 2008 - Baccarelli, A., S.M. Giacomini, C. Corbetta, M.T. Landi, M. Bonzini, D. Consonni, P Grillo, D.G. Pattterson, Jr., A.C. Pesatori, and P.A. Bertazzi, 2008. Neonatal thyroid function in Seveso 25 years after maternal exposure to dioxin. *PLoS Med* 5:e161. 44

USEPA, 2012a. Bioavailability. U.S. Environmental Agency, Region 8. Available at: http://www.epa.gov/region8/r8risk/hh_rba.html#recs.

USEPA, 2012b. Generic Tables. Available online at: http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/Generic_Tables/index.htm. Last updated May 20, 2012. U.S. Environmental Protection Agency, Mid-Atlantic Risk Assessment.

USEPA, 2012c. Integrated Risk Information System (IRIS). U.S. Environmental Protection Agency. Available online at: http://www.epa.gov/ncea/iris/

USFWS, 1982. Life History Requirements of Selected Finfish and Shellfish in Mississippi Sound and Adjacent Areas. FWS/OBS-81/51. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC. March.

USFWS, 1983. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Gulf of Mexico), Sea Catfish and Gafftopsail Catfish. FWS/OBS-82/11.5. TR EL-82-4. U.S. Fish and Wildlife Service. October.

USFWS, 2006. 2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation: Texas. FHW/06-TX. U.S. Fish and Wildlife Service. May.

Vamvakas, A., J. Keller, and M. Dufresne, 1996. *In vitro* induction of CYP 11-associated activities in human and rodent cell lines by commercial and tissue-extracted halogenated aromatic hydrocarbons. *Environ. Toxicol. Chem*. 15(6):814-823.

Van den Berg, M., L.S. Birnbaum, M. Denison, M. DeVito, W. Farland, M. Feeley, H. Fiedler, H. Hakansson, A. Hanberg, L. Haws, M. Rose, S. Safe, D. Schrenk, C. Tohyama, A. Tritscher, J. Tuomisto, M. Tysklind, N. Walker, R. E. Peterson, 2006. The 2005 World Health Organization Reevaluation of Human and Mammalian Toxic Equivalency Factors for Dioxins and Dioxin-Like Compounds. *Toxicol. Sci.* 93(2):223-241.

Walker, N.J., P.W. Crockett, A. Nyska, A.E. Brix, M.P. Jokinen, D.M. Sells, J.R. Hailey, M. Easterling, J.K. Haseman, M. Yin, M.E. Wyde, J.R. Bucher, and C.J. Portier, 2005. Dose-additive carcinogenicity of a defined mixture of "dioxin-like compounds." *Environ. Health Perspect.* 113(1):43-48.

Wang, I-C, and W Lee, 2010. Polychlorinated dibenzo-p-dioxin, polychlorinated dibenzofurans and polychlorinated biphenyls in farmed fish, water, sediment, and feed. *J. Environ. Sci. Health. Part A: Toxic/Hazardous Subst Environ. Eng.* 45(2):201-210.

WHO, 1991. Summary Report – Consultation on Tolerable Daily Intake from Food of PCDDs and PCDFs. World Health Organization, Bilthoven, the Netherlands. December 1990, EUR/ICP/PCS 030(S) 0369n, World Health Organization, Regional Office for Europe, Copenhagen.

WHO, 1992. Tolerable daily intake of PCDDs and PCDFs. *Toxic Substances Journal* 12:101-128.

WHO, 1998. Assessment of the health risk of dioxins: reevaluation of the tolerable daily intake (TDI). World Health Organization European Centre for Environment and Health, International Programme on Chemical Safety.

Wilson, N.D., N.M. Shear, D.J. Paustenbach, and P.S. Price, 1998. The effect of cooking practices on the concentration of DDT and PCB compounds in the edible tissue of fish. *J. Expos. Anal. Epidemiol.* 8:423–440.

Winward, 2007. Lower Duwamish Waterway Remedial Investigation. For submittal to U.S. Environmental Protection Agency, Seattle, WA and Washington State Department of Ecology, Bellevue, WA. November 12, 2007.

Wiebel, F.J., M. Wegenke, and F. Kiefer, 1996. Bioassay for determining 2,3,7,8-tetrachlorodibenzo-pdioxin equivalents (TEs) in human hepatoma HepG2 cells. *Toxicol. Lett.* 88(1-3):335-338.

Xu, L., A.P. Li, D.L. Kaminski, and M.F. Ruh, 2000. 2,3,7,8 Tetrachlorodibenzo-*p*-dioxin induction of cytochrome P4501 in cultured rat and human hepatocytes. *Chem. Biol. Interact.* 124(3):173-189.

Yanez-Arancibia, A., and A. Lara-Dominguez, 1988. Ecology of three sea catfishes (*Ariidae*) in a tropical coastal ecosystem – Southern Gulf of Mexico. *Mar. Ecol. Prog. Ser.* 49:215-230.

Zeiger, M., R. Haag, J. Hockel, D. Schrenk, and H.J. Schmitz, 2001. Inducing effects of dioxin-like polychlorinated biphenyls on CYP1 in the human hepatoblastoma cell line HepG2, the rat hepatoma cell line H4IIE, and rat primary hepatocytes: Comparison of relative potencies. *Toxicol. Sci.* 63(1):65-73.